EFFECTS OF ELEVATED SEDIMENTS LEVELS FROM PLACER MINING ON SURVIVAL AND BEHAVIOR OF IMMATURE ARCTIC GRAYLING

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ABSTRACT

The effect of placer mining effluentson Arctic grayling (*Thymallus arcticus*) fingerling and egg survival was tested in mined and unmined streams in interior Alaska. Also the influence of turbidity on Arctic grayling reactive distance and avoidance behavior was tested in a laboratory choice chamber.

Arctic grayling fingerlings suffered less than 1% mortality during a 96-hr toxicity test in both clear (mean NTU = 1.4) and mined (mean NTU = 445) streams. Arctic grayling eggs did not show significantly ($p \geq 0.1$) higher mortality in mined streams than in unmined streams. In a laboratory choice chamber test, Arctic grayling avoided water with a turbidity above 20 NTU (nephelometric turbidity units). Arctic grayling reactive distance diminished proportional to the natural logarithm of turbidity.

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INTRODUCTION

The effects of placer mining pollution on Arctic grayling (*Thymallus arcticus*) are not easily determined using conventional techniques. Standard methods, such as toxicity tests for lethal effects, show that placer mining effluents in concentrations that usually occur below mining sites is not acutely toxic to most life stages of Arctic grayling. Reynolds et al. (1988) reported that the only life stage that suffered lethal effects was sac-fry. Prior to the present study, no work had been conducted to test the effects of placer mining pollution on Arctic grayling eggs.

Although placer mining pollution may not cause death to juvenile and adult Arctic grayling during the period of time toxicity tests are conducted, this does not necessarily imply that placer mining pollution has no effect on Arctic grayling.

Various researchers have shown that the habitat in which Arctic grayling live is profoundly affected by placer mining pollution. Stream morphology is altered through pool filling and bed sealing (Bjerklie and LaPerriere 1985). Primary production, and invertebrate abundance and diversity are greatly decreased (Wagener and LaPerriere 1985, Van Nieuwenhuyse and LaPerriere 1986). These findings indicate that placer mining pollution does have an effect on Arctic grayling by affecting their habitat. In recent years, the Alaska Cooperative Fishery Research Unit has emphasized research on the more direct effects of discharge sediments on survival and behavior of Arctic grayling.

OBJECTIVES

The objectives of my research project were to determine the effects of placer mining pollution on Arctic grayling survival and behavior. Specifically:

1) whether the Arctic grayling egg or fingerling life stage is sensitive to placer mining pollution; and

2) whether increased turbidity results in behavioral changes, such as avoidance of turbid water or changes in feeding behavior, resulting from the grayling's reduced ability to detect and react to food in turbid water.

This thesis is divided into two sections. The first section reports on toxicity studies, the second on behavioral studies.

BACKGROUND

The experiments conducted in this study investigated specific aspects of placer mining pollution and specific life stages of Arctic grayling. This section introduces background information on Arctic grayling life history and on the placer mining industry.

SPORT VALUE

Arctic grayling is an important Alaska sport fish. Arctic grayling is the most popular sport fish among anglers fishing in interior Alaska (Holmes 1981). In 1980, 170,137 Arctic grayling were harvested by anglers (Mills 1981). Only pink salmon (*Oncorhynchus gorbuscha*) and Dolly Varden (*Salvelinus malma*) plus Arctic char (*Salvelinus alpinus*) were caught by sport anglers in greater numbers than Arctic grayling in 1980 in Alaska (Armstrong 1986).

LIFE HISTORY

The Arctic grayling occurs from Hudson Bay west to the Kara and Ob rivers of Northern Asia. Arctic grayling occur throughout northwest Canada and Alaska. The southern range of Arctic grayling extended to rivers flowing into Lakes Michigan, Huron and Superior in northern Michigan until about 1936. Currently Arctic grayling occur in the headwaters of the Missouri River above Great Falls, Montana, and Arctic grayling have been introduced into the mountainous areas of Vermont, Utah, and Colorado (Scott and Crossman 1973).

Reproduction

The spawning period for Arctic grayling in Alaska ranges from late April to early July, with most Arctic grayling spawning between mid-May and mid-June (Wojcik 1954, Warner 1955, Schallock 1966, Roguski 1967, Roguski and Tack 1970, de Bruyn and McCart 1974, Tack 1974, Bendock 1979). Rising water temperature and spring flooding may trigger Arctic grayling spawning. Grayling have been found to spawn in 4 °C water (Warner 1955, Tack 1973), and in the turbid conditions of spring flood in the Chatanika River near Fairbanks(Schallock 1965).

Arctic grayling spawn in main rivers, large and small tributaries to rivers and lakes, intermittent streams, and lakes (Warner 1957, Bendock 1979). In rivers and streams, Arctic grayling usually spawn in riffle areas of pea-sized gravel (Warner 1955, Tack 1971). Tack (1973) found that most spawning occurred over gravel between 0.075 and 38.1 mm. Nelson (1954) found Arctic grayling eggs present in gravel or rubble riffles, and absent in sand or silt pools, in Montana streams. Tack (1971) found that male Arctic grayling defend a spawning territory about 2-2.5 m wide by 2.5-3 m long with an average water depth of 0.3 m (range: 0.2-0.7 m), and an average water velocity of 0.8 m/s (range: 0.3-1.5 m/s). Grayling eggs are deposited at a depth of about 25 mm into the gravel (Kratt and Smith 1977). The eggs hatch in about 186 degree days (Kratt and Smith 1977). Degree days are equal to a number of days times the average temperature (°C) over those days. The fry absorb their yolk sacs and leave the gravel 3-4 days after hatching.

Young-of-the-Year

After emerging from the stream gravel, young Arctic grayling live in areas of quiet water such as between rocks at the lower end of gravel bars (Vascotto 1970), backwaters, and side channels (Tack 1971; McCart, Craig, and Bain 1972; de Bruyn and McCart 1974). They remain, often schooled, in these quiet areas until late summer when they become territorial and move into deeper water (Vascotto 1970, de Bruyn and McCart 1974). In early fall they may leave the smaller streams and enter larger rivers, lakes, or spring-fed areas for overwintering (Tack 1980).

Migrations

Interior Alaska Arctic grayling may migrate up to 160 km to reach overwintering sites (Tack 1980). In larger rivers of interior Alaska, such as the Chena, Goodpaster, and the Chatanika, most Arctic grayling inhabiting the upper reaches and tributaries migrate downstream to overwinter in the deeper water areas of the main stem (Tack 1980). Although this is the commonly accepted migrational pattern, I have observed Arctic grayling moving into the upper reaches of Twelvemile Creek at freeze-up. In 1987 and 1988, Arctic grayling were observed overwintering in an aufeis covered area of upper Twelvemile Creek (A. Burkholder, Alaska Cooperative Fishery Research Unit, University of Alaska., pers. com. 1988). This suggests that Arctic grayling use some headwater areas for overwintering.

Food and Feeding Habits

Grayling feed primarily on insects that are drifting in the water column or floating on the water surface (Vascotto 1970, Vascotto and Morrow 1973). During June and early July, in the Delta Clearwater River, Arctic grayling feed on a 24-hr basis, and later in the year they only stop feeding at darkness (Reed 1964). Among 1,300 Arctic grayling collected by Reed (1964) from 13 watersheds in Alaska, none were found to have empty stomachs. This suggests that, during summer at least, Arctic grayling are active feeders.

PLACER MINING

In the early 1890's gold miners traveled down the Yukon River to find gold in the Central area, about 100 miles Northeast of Fairbanks. And gold they found. The Central Mining District soon became, and has remained, one of Alaska's most active gold mining areas (Bundtzen et al. 1984)

Geology of Placer Mining

Gold is denser than most other sediments. Over eons of time as streams wash sand and gravel from the mountains into the valleys, the bits of gold tend to settle into a deposit of gravel closest to bedrock. This is called the placer deposit, and it consists of ancient gravels often containing enough fine mesh gold to be worth mining.

Placer deposits are usually located in stream valleys. They may be close to the surface or they may be covered by many meters of more recent deposits called overburden.

The basic processes of placer mining involve removing the overburden, excavating the placer deposit (pay dirt), and separating the gold from the bulk of the deposit. Pollution abatement and site reclamation are recent additions to the basic processes.

Historically, several systems of mining evolved to extract placer gold. These include panning, drift mining, dredging, hydraulic mining, and open cut mining. Of these, open cut mining is the major method used in interior Alaska today.

During open cut placer mining large earth moving equipment, including bulldozers, backhoes, and draglines, is commonly used to remove vegetation, topsoil and gravel overburden above the placer deposit. This first step exposes streams to increased sediment pollution from erosion of the exposed soil. Then placer gravels are flushed through a sluice box (an artificial stream) where they are washed with water. The heavier gold settles out and collects on the riffles, while the coarser gravels and lighter fines wash away.

Wash water from the sluice box is a primary source of placer mining pollution. This water contains small particles of dirt and sand (settleable solids) that settle out when the water is held in a quiescent pond. The water also contains very fine particles that would require many hours or days, or the help of a settling agent, to settle out of the water. The particulates may carry, or be associated with, high levels of heavy metals or other pollutants (such as arsenic).

Placer Mining Pollution

Placer mining operations often release sediments, heavy metals, and turbid water into streams that are inhabited by Arctic grayling. The effects of placer mining pollution on downstream water quality and aquatic habitat have been investigated by researchers and resource agencies in Alaska (e.g. Chang 1979, Yang 1979, Madison 1981, Bjerklie and LaPerriere 1985, Wagener and LaPerriere 1985, Weber and Post 1985, Mack 1986, Van Nieuwenhuyse and LaPerriere 1986, Lloyd 1987). These studies have documented an often dramatic increase in turbidity, suspended solids, and total metals in waters affected by placer mining pollution.

Increased sediment loads alter channel morphology by sediment deposition (Dames and Moore 1986), limit surface to subsurface water exchange (Bjerklie and LaPerriere 1985), and decrease the average particle size of the stream bottom substrate (Weber and Post 1985).

Biological effects of increased sediment levels include loss of habitat for fish and other aquatic taxa (Weber and Post 1985, Dames and Moore 1986), decreased survival of fish in the sac-fry stage (Reynolds et al. 1988), decreased populations of aquatic macroinvertebrates and alterations in macroinvertebrates community structure (Wagener and LaPerriere 1985, Weber and Post 1985, Weber 1986), and decreased algal production (Van Nieuwenhuyse and LaPerriere 1985, Lloyd et al. 1987).

TOXICITY STUDIES

INTRODUCTION

The object of this study was to determine the sensitivity of two Arctic grayling life stages to placer mining pollution in streams in interior Alaska. Adult Arctic grayling appear to be more tolerant of placer mining pollutants than younger life stages (Simmons 1984, Reynolds et al. 1988). Little is known about the tolerance of Arctic grayling eggs to placer mining pollutants. This component of the study tested the effects of placer mining pollution on Arctic grayling egg and fingerling life stages.

Sensitivity of Arctic grayling to placer mining pollution

Simmons (1984) reported that Arctic grayling caged in streams carrying placer mining sediments suffered gill damage, dietary deficiencies, and slowed maturation.

Reynolds et al. (1988) conducted toxicity tests with 2200 Arctic grayling sac-fry in Birch (mined) and Twelvemile (unmined) creeks. The mean survival of sac-fry in the mined stream was 85%, 74%, 59%, and 53% in 24, 48, 72, and 96-hr respectively, while the mean survival of sac-fry in the unmined stream was 94%, 86%, 85%, and 87% in 24, 48, 72, and 96-hr respectively. Arctic grayling sac-fry had significantly higher mortality in mined Birch Creek than sac-fry in unmined Twelvemile Creek.

SENSITIVITY OF ARCTIC GRAYLING FINGERLINGS TO PLACER MINING POLLUTION

Methods

A 96-hr toxicity test using Arctic grayling fingerlings (about 45 mm fork-length) was conducted in Twelvemile and Birch creeks from July 30 through August 1, 1986. The Arctic grayling were obtained from Clear Hatchery, Alaska Department of Fish and Game, and were transported to the study site in plastic bags with oxygen gas, refrigerated with ice in plastic coolers. During the test the Arctic grayling were kept in 19-liter plastic buckets that had six screened (1 mm mesh) holes (65 mm diameter) in them to allow stream water to pass through. Twelve buckets were placed in Birch Creek (mined) and twelve buckets were placed in Twelvemile Creek (unmined). One hundred Arctic grayling fingerlings were placed in each bucket. A total of 2400 Arctic grayling were used.

During the 96-hr toxicity test, stream conditions were monitored every 6 hr. Measurements taken included turbidity, (NTU) using a Hach(R) model 16800 portalab nephelometric turbidimeter; settleable solids, using an Imhoff cone; and total suspended solids (TSS) and total solids (TS), by filtering, drying and weighing; all procedures followed standard methods (APHA 1980). Dissolved solids (DS), the portion of total solids that passed through a glass fiber filter, were calculated by subtracting TSS from TS.

Grab samples of water for the determination of heavy metals concentrations were collected at the beginning, middle, and end of the 96-hr test in acid washed Nalgene(R) bottles. The samples were acidified on site, to a pH of less than 2, with ultra-pure nitric acid. Analyses of arsenic, copper, lead, and zinc were done by Northern Testing Laboratories, Inc., Fairbanks, Alaska, using EPA method number 4.1.4 (total recoverable metals) (USEPA 1979).

Results

The Arctic grayling in the mined stream were exposed to significantly ($p < 0.01$) higher turbidity than the Arctic grayling in the unmined stream. During the test, the mean turbidity in Twelvemile Creek was 1.4 NTU, (standard deviation (SD) = 0.5) while the mean turbidity in Birch Creek was 445 NTU (SD = 147).

Total solids (TS) in Twelvemile Creek averaged 100 mg/L (SD = 0), while the TS in Birch Creek averaged 500 mg/L (SD = 141). Total suspended solids (TSS) in Twelvemile Creek averaged 25 mg/L (SD = 35), while TSS in Birch Creek averaged 375 mg/L (SD = 176). DS in Twelvemile Creek averaged 75 mg/L (SD = 35), while the DS in Birch Creek averaged 125 mg/L (SD = 35).

The fingerlings in the mined stream were exposed to significantly (Wilcoxon Rank Sum W test) higher concentrations of each of the four metals tested than were fingerlings in the unmined stream. Concentrations of total recoverable heavy metals in mined Birch Creek were 3 - 9 times higher than in unmined Twelvemile Creek (Table 1).

In Twelvemile Creek, 8 of 1200 Arctic grayling fingerlings died in 96-hrs, while in Birch Creek 4 of 1200 Arctic grayling fingerlings died in 96-hrs.

Discussion

Turbidity levels averaging about 418 NTU and TSS levels averaging about 500 mg/L, resulting from placer mining, had no significant effect on the mortality rate of Arctic grayling fingerlings over a four day period.

Table 1. Total recoverable metals concentrations in mined Birch and unmined Twelvemile creeks, July 30, - August 1, 1986.

metal	median concentration		significance	
	mg/L	mg/L	p	N
	Birch Cr.	Twelvemile Cr.		
zinc	0.039	0.008	0.013	4
arsenic	0.009	0.001	0.008	9
copper	0.030	0.010	0.015	3
lead	0.004	0.001	0.017	4

SENSITIVITY OF ARCTIC GRAYLING EGGS TO PLACER MINING POLLUTION

The objective of this component of the study was to test the null hypothesis that Arctic grayling eggs, in modified Vibert egg boxes, placed in clear water streams will exhibit the same rates of hatching success as identically treated Arctic grayling eggs placed in streams receiving sediment from placer mining.

Description of Study Sites

Streams in the Birch Creek watershed (Figure 1) have historically supported populations of Arctic grayling. Weber and Post (1985) report that local residents in Central fished in Miller and Mastodon creeks (currently mined tributaries to Mammoth Creek) in the early 1980's, and miners operating on Porcupine Creek (a mined tributary to Crooked Creek) and Bonanza Creek (a mined tributary to Porcupine Creek) found Arctic grayling in these waters in the late 1970's and early 1980 "before all the miners moved into upper Bonanza Creek." Weber and Post (1985) also report that employees at the U.S. Bureau of Land Management summer fire camp at Central reported successful sportfishing for Arctic grayling in Crooked Creek near the BLM station until 1977. Miners, residents of Central, and other rural residents reported "good sportfishing success" for Arctic grayling in Ptarmigan Creek. Historically, Deadwood Creek, Mammoth Creek and its tributaries, Crooked Creek, and Birch Creek and its tributaries (Ptarmigan, Golddust, Eagle, and Butte creeks) have all supported Arctic grayling (Weber and Post 1985, Weber, pers. comm.).

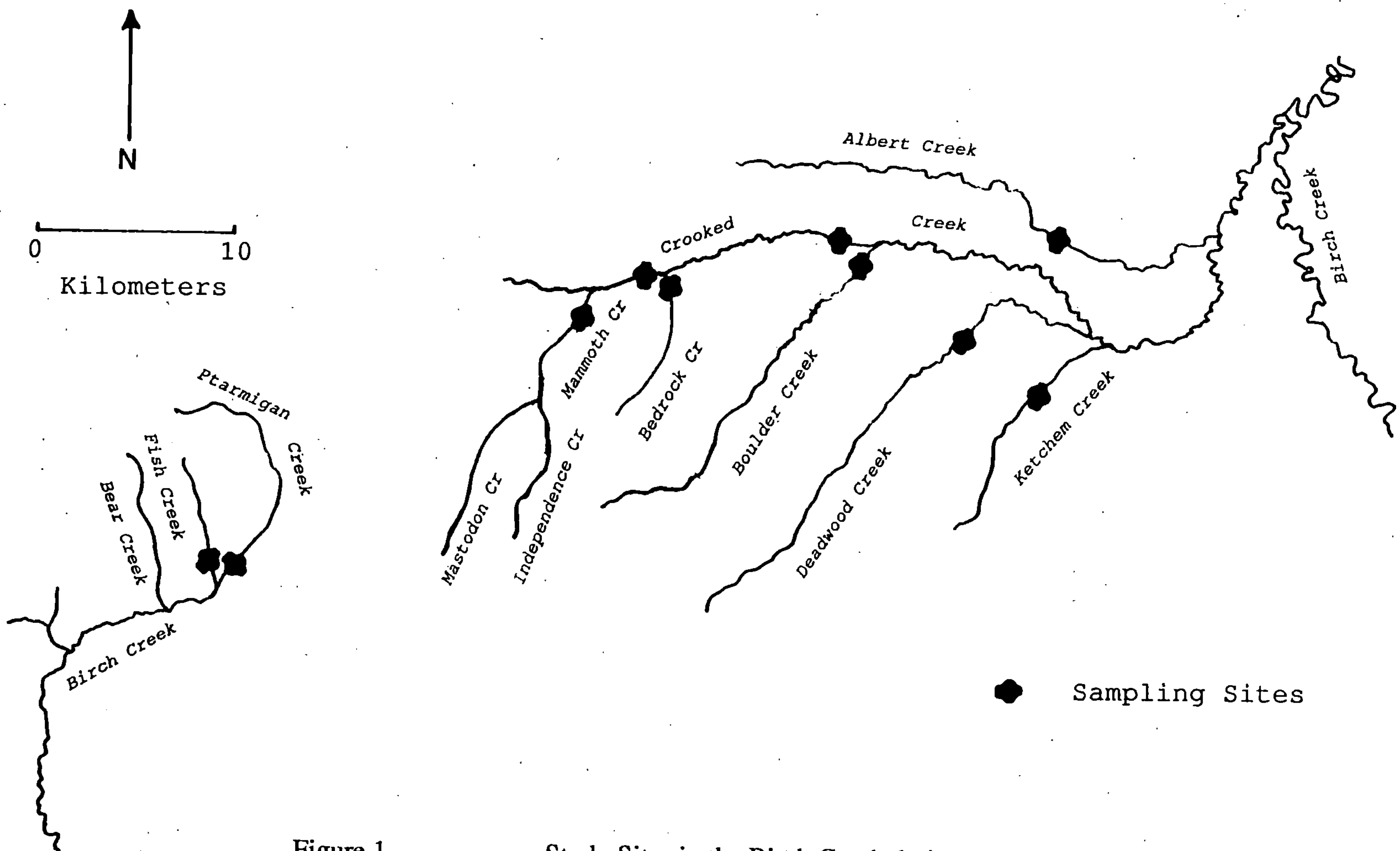

Figure 1. Study Sites in the Birch Creek drainage

Ketchem Creek becomes a braided, intermittent stream below Circle Hot Springs Road, and may prohibit fish passage into the upper reaches.

The Birch Creek watershed contains populations of Arctic grayling; slimy sculpin (*Cottus cognatus*); broad, humpback, and round whitefish (*Coregonus nasus, C. pidschian,* and *Prosopium cylindraceum,* respectively); northern pike (*Esox lucius*); burbot (*Lota lota*); and Dolly Varden char (*Salvelinus malma*). Of these species, Arctic grayling and slimy sculpin are the most common. Char, pike, burbot, and whitefish occur primarily in the lower reaches of Crooked Creek and Birch Creek; sheefish (*Stenodus leucichythys*), chinook, coho, and chum salmon (*Oncorhynchus tshawytscha, O. kisutch,* and *O. keta,* respectively) occur in the lower reaches of Birch Creek (Alaska Department of Fish and Game 1987).

Arctic grayling have occurred historically throughout the Goldstream Creek drainage (Figure 2). Local residents reported fishing successfully for Arctic grayling in Goldstream Creek near Murphy Dome Road until 1986, near Goldstream Road in the mid 1970's before the resurgence of placer mining and in the upper reaches of Goldstream Creek until the mid-1970's (Weber 1986). Turbidity levels in Goldstream Creek were generally below 30 JTU (roughly equivalent to NTU) in 1970-71 (LaPerriere and Nyquist 1973). In May and June of 1984 turbidity averaged 315 NTU and ranged from 96 to 1560 NTU (Weber and Robus 1987).

Arctic grayling have been recently collected (1984-1987) in Albert, Bedrock, Boulder, McManus, and Faith creeks, and the upper reaches of Birch Creek (near the confluence of Twelvemile Creek) (Weber and Post 1985, Dames and Moore 1986). In the Goldstream Creek drainage, Arctic grayling have been collected in the lower

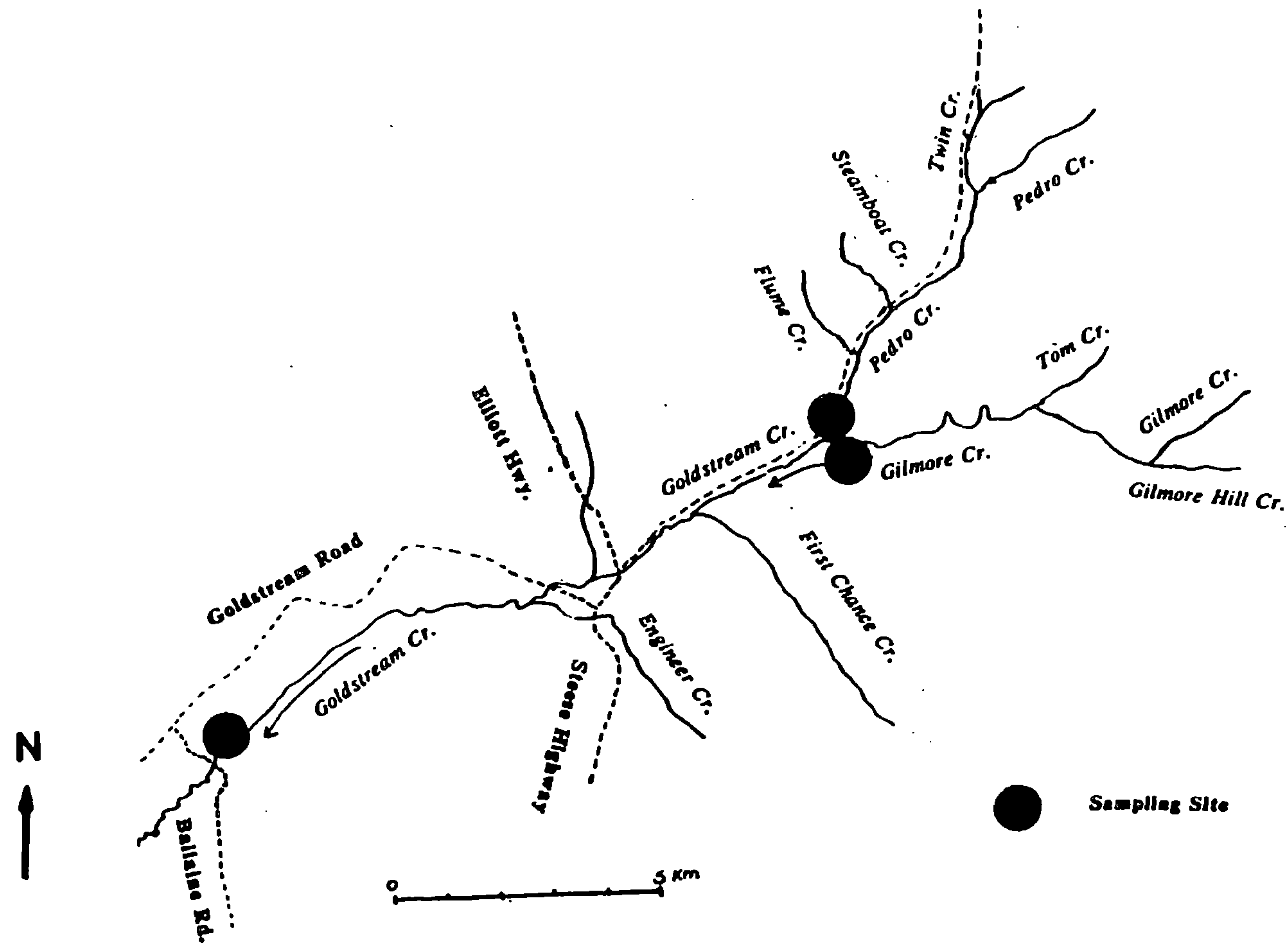

Figure 2. Sample sites in the Goldstream Creek drainage.

reaches of Goldstream Creek (about 10 km above the confluence with the Chatanika River) and in Flume Creek, a tributary to Pedro Creek (Weber and Robus 1987).

Finally, Arctic grayling are common in Faith and McManus creeks in the Chatanika River drainage (Vascotto 1970). An Arctic grayling toxicity test site was located in Faith Creek and another was located in McManus Creek. Both test sites were near the confluence of the two creeks (Figure 3).

Physical Habitat Features

Study streams in the Crooked Creek, Birch Creek, Goldstream Creek, and Faith Creek and McManus Creek drainages were relatively small second- and third-order tributaries. These streams were typically of moderate gradient (0.5 to 3%), with unaltered streams having relatively straight channels with short meanders. Stream channels that had undergone placer mining were almost entirely straight.

Aufeis (overflow ice) and anchor ice (ice on the stream bed) were common in nearly all of the streams sampled, in late May of 1987. Except during break-up and high rainfall events, the study streams typically have discharges that ranged from 0.06 to 1.1 m^3/s (2 to 40 ft^3/s) (Mack 1986). A range of physical habitat features of aspect, gradient, drainage area, channel shape, elevation of the headwaters, and average discharge were represented in mined and unmined streams. The physical characteristics of each study stream are summarized in Table 2, based on information in Mack (1986), and Bjerklie and LaPerriere (1985). Abbreviations used for site locations are in the appendix (Table 7).

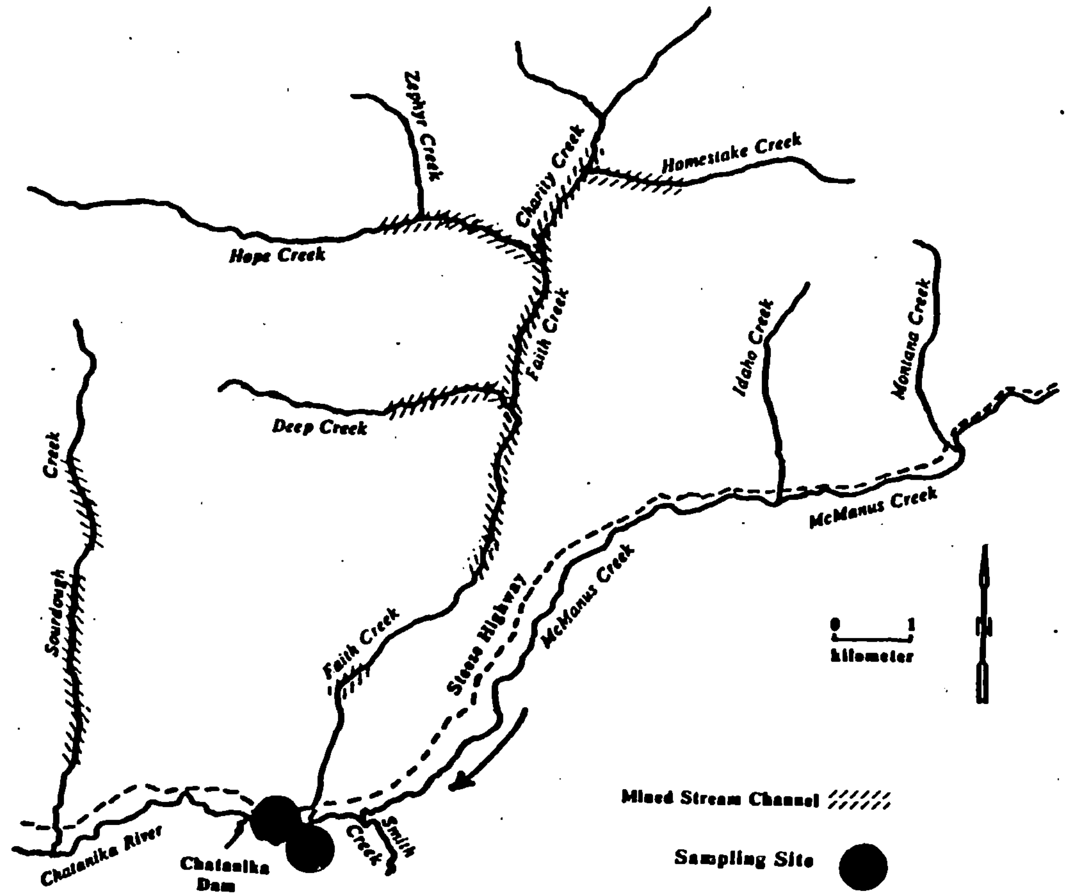

Figure 3. Sample sites on Faith Creek and McManus Creek.

Table 2. Physical characteristics of study streams (Mack 1986, Bjerklie and LaPerriere 1985).

Stream	Drainage Area (km^2)	Channel Length (km)	Elev of Head waters (m)	Avg. Slope (%)	Aspect (deg)	Avg. Disch. (m^3/s)
Faith	156	-	-	-	-	2.7
McManus	201	-	-	-	-	2.4
Deadwood	91.2	21.6	1065	2.0	37	0.3
Ketchum	31.8	9.7	625	4.0	30	0.09
Boulder	85.5	22.0	1050	2.0	40	-
Crooked	432.5	19.5	280	0.8	100	1.1
Bedrock	25.4	9.7	1250	6.0	20	0.06
Mammoth	107.5	6.6	480	2.0	30	0.58
Fish	19.2	7.7	1250	5.0	170	-
Birch at Twelvemile Confl.	231	13.0	695	0.7	-	4.2

Methods

This section is divided into two parts. The first part describes methods used to measure water quality characteristics, including turbidity, total suspended solids, settleable solids, pH, metals, and temperature. The second part describes methods for incubating Arctic Grayling eggs and testing toxic responses.

Water Quality

Turbidity was measured in each study stream 3 - 17 times during the experiment using a Hach(R) model 16800 portalab turbidimeter. The early samples were taken before mining began that spring.

Total suspended solids samples were taken 2 - 17 times during the experiment. They were either taken as grab-samples with a Nalgene(R) bottle, or with an ISCO(R) sampler. An ISCO(R) sampler is an automated water sampler that was programmed to collect four water samples each day for the duration of the study. Samples collected within a given day were combined to give a daily average. The water samples were transported back to the laboratory in a cooler with ice. There they were refrigerated at 4° C until analyzed.

The smallest marked measure on the Imhoff cones I used in this study was 0.5 mL/L. I could extrapolate below this measure to 0.4 mL/L, but no further because a plastic screw cap obscured the rest of the end of the cone except for the base. Through the base of the cone I was able to observe trace amounts (a few grains on the bottom), and no detectable amount (zero settleable solids).

Therefore, the minimum settleable solids levels reported were 0.4 mL/L, less than 0.4 mL/L, trace, or no detectable amount (ND), except for settleable solids measurements from Pedro, Gilmore and Goldstream creeks which were taken by the Alaska Department of Fish and Game (ADF&G) with an Imhoff cone that measured to 0.2 mL/L. ADF&G reported minimum settleable solids measurements as less that 0.2mL/L, trace, and ND.

On May 22-24, and June 23, grab samples of stream water were taken at each of the egg-toxicity-test sites using acid-washed Nalgene(R) bottles. Samples were preserved by adjusting the pH of the sample to below 2 with ultrapure concentrated nitric acid. The total recoverable form of zinc, arsenic, lead, copper, gold, and silver were analyzed by Northern Testing Laboratories, Inc (NTL) using EPA method number 4.1.4 (total recoverable metals). Two heavy metals check samples for arsenic, copper, lead, and zinc; and two heavy metals check samples for silver, from the EPA Environmental Monitoring and Support Laboratory, were submitted with the water samples to NTL. NTL values for arsenic, copper, and lead fell within the 95% confidence limits provided by the EPA; however, the laboratory's values for zinc were above the 95% confidence limits, and the values for silver fell below the 95% confidence limits. These biases should not effect our hypotheses, which concern relative metals concentrations, but the biases should be considered when looking at the absolute metals concentrations. The EPA did not provide a check sample for gold, and no quality control work was done on this metal.

The results of trace metals analyses were compared with EPA water quality criteria. The criteria used were the Criterion Maximum Concentration (CMC), which is equal to one-half the Final Acute Value; and the Criterion Continuous Concentration (CCC), which is equal to the lowest of the Final Chronic Value, the Final Plant Value, and the Final Residue Value, unless other data show that a lower value should be used (EPA 1986).

In addition, pH was measured with a Hach(R) model 19000 digital pH kit, and temperature was measured with a hand held mercury thermometer.

Toxicity Tests

I incubated Arctic grayling eggs in mined and unmined streams and observed their survival. The Arctic grayling eggs were acquired through Clear Hatchery, a local state-operated (ADF&G) fish hatchery. Clear Hatchery's eggs are collected from tested sources including Jack Lake, Moose Lake, and Goodpaster River. Arctic grayling eggs are 2.7 -4.3 mm in size (Scott and Crossman 1973).

Arctic grayling eggs are fairly hardy during the first 24 hr of development (D. Parks Hatchery Manager, Clear Hatchery, Clear, Alaska, pers. com. 1987). They then become too sensitive to handle until they reach the "eyed stage." All eggs used in this study were placed in site positions within 24 hr of when they were stripped.

The eggs were kept cool and were handled as quickly as possible. They were counted into Whirl Paks(R) (fifty eggs per pack) at a laboratory at the university and were promptly transported, refrigerated on ice in insulated coolers, to the field site. At the study site, substrate from the site was added to the egg box so that the eggs, when added, were buried about 25 mm deep.

The eggs boxes were made of Nitex monofilament nylon screening designed after Vibert egg boxes (Vibert 1959). The screening had an open area of 45%, and a mesh opening of 710 microns. This allowed passage of water while preventing passage of eggs or sac-fry, as required by my State of Alaska Department of Fish and Game fish transport permit. Each egg box was 14 cm by 10 cm with a 3.5 cm depth.

Egg boxes were sealed by folding the upper edge of the box over a piece of Tygon(R) tubing, then a split piece of firm PVC tubing was slipped over the Tygon(R) to complete the seal. Three egg boxes were tied to a piece of steel rod and gently placed in the stream. Three pieces of steel rod were previously placed at each site, so there were initially nine egg boxes per site. Streams were selected to represent a range of water conditions that varied from clear water to slightly turbid to very turbid water. Sixteen sites were studied: five sites were in clear water streams, and eleven sites were in streams that had been mined in the previous season.

Experimental sites in the streams were chosen randomly by generating pairs of random numbers and using them as X-Y coordinates on an imaginary grid laid over the stream, with one walking pace used as the unit of measure. Egg bags were placed in the nearest quiescent area that did not contain anchor ice.

Spatial controls were used by pairing clear streams with otherwise similar streams effected by placer mining pollution. This follows the paired-watershed approach of Bates and Henry (1928).

The estimated duration of the study was two to four weeks, because Arctic grayling eggs hatch in 13 - 18 days at 7 - 11 $^{\circ}$C (Scott and Crossman 1973). In addition to the eggs used as subjects of this study, a separate set of observation egg bags was incubated and observed frequently to determine the best times to examine the eggs in the study. I desired to examine the eggs after the dead eggs had decomposed enough to be clearly discernible from the living eggs, yet before the dead eggs had decomposed beyond recognition. Egg development and decomposition were initially slow due to low water temperature, but both increased in the latter part of the study. In accordance with the development and decomposition of the observation eggs, three egg bags were removed from each site on May 30, June 3, and June 12, 1987. At these times the number of living eggs was recorded and all the eggs and larvae were counted and preserved in a mixture of 75 parts 50% ethanol, 25 parts glacial acetic acid, and 2-3 mL of formalin (J. Sullivan, Microbiologist, Alaska Department of Fish and Game, Division of Fisheries Enhancement, Rehabilitation and Development, Anchorage, Alaska. pers. com. 1987).

Results

The results of turbidity, total suspended solids, settleable solids, pH and temperature, heavy metals concentrations, and egg survival monitoring are summarized here. Actual values are presented in the Appendix (Table 8).

Water Quality

Arctic grayling eggs were exposed to significantly higher turbidities in the mined streams than in the unmined streams ($p < 0.01$), (Table 3). Stream water turbidity measured throughout the sampling period ranged from 2 NTU to 18 NTU in the

Table 3. Stream water turbidity at study sites during the incubtion periods of three sets of eggs. Turbidity is measured in NTU, N is the total number of measurements taken during the study.

SITE	Avg. May 21-May 30	Avg. May 21-June 3	Avg. May 21-June 13	Min.	Max.	N
CLEAR SITES						
ALBERT	10	8	7	3	14	4
BEDROC	5	5	4	2	6	6
BOLDER	13	13	13	11	14	3
FISH	4	9	9	3	18	3
MCMANS	6	4	4	2	6	5
Average, all clear sites	8	8	7			
MINED SITES						
BIRFSH	320	568	568	250	950	4
CRKBLD	83	65	65	29	85	3
CRKBLM	127	135	264	125	650	4
CRKBRK	77	115	140	33	295	6
DEDWOD	88	73	60	11	120	5
FAITH	88	65	63	21	120	5
KETCHM	98	121	339	61	1200	6
MAMOTH	300	300	347	275	440	3
PEDRO	18.8	18.9	18.4	10.2	27.7	15
GILMORE	22.9	22.9	22.9	19.6	28.2	4
GOLDSTREAM	39.7	45.7	42	15.2	105	17
Average, all mined sites	115	139	175			

unmined sites, with an average for all unmined sites of 7 NTU. Stream water turbidities in mined sites were nearly 25 times higher, with an average of 175 NTU (turbidities ranged from 10.2 NTU, a one point measurement in Pedro Creek, to 1200 NTU, measured in Ketchem Creek). Of the mined streams, Birch Creek at Fish Creek had the highest average turbidity, (568 NTU with a range of 250 to 950 NTU).

Arctic grayling eggs in unmined streams were consistently exposed to lower turbidities than Arctic grayling eggs in mined sites. Stream water turbidity in mined sites averaged 14 times higher than in unmined sites from the first day until the first set of egg bags was removed, 17 times higher from the first day until second set of egg bags was removed, and 25 times higher over the duration of the test (Figure 4).

Total suspended solids (the solids that remain on a glass fiber filter with a pore size of 0.3 *um*) levels averaged about 11 times higher in mined streams than in unmined streams (Table 4). TSS concentrations were low in unmined streams except during high rainfall events and during breakup. For example, the TSS concentration in Albert Creek, an unmined stream that is usually organically stained, was 290 mg/L during a high water event that occurred on June 12. This level is about 3.5 times higher than the average TSS concentration measured in Albert Creek. The grand mean of TSS concentration in all unmined streams was 44 mg/L (range = 0 to 290 mg/L, n = 18).

TSS concentrations were consistently higher in mined streams than in unmined streams. The average TSS measured over the sampling period was 505 mg/L, with a range of 14 to 2301 mg/L (n = 37). TSS levels peaked in mined streams during the same rainfall event that caused elevated TSS levels in Albert Creek. For

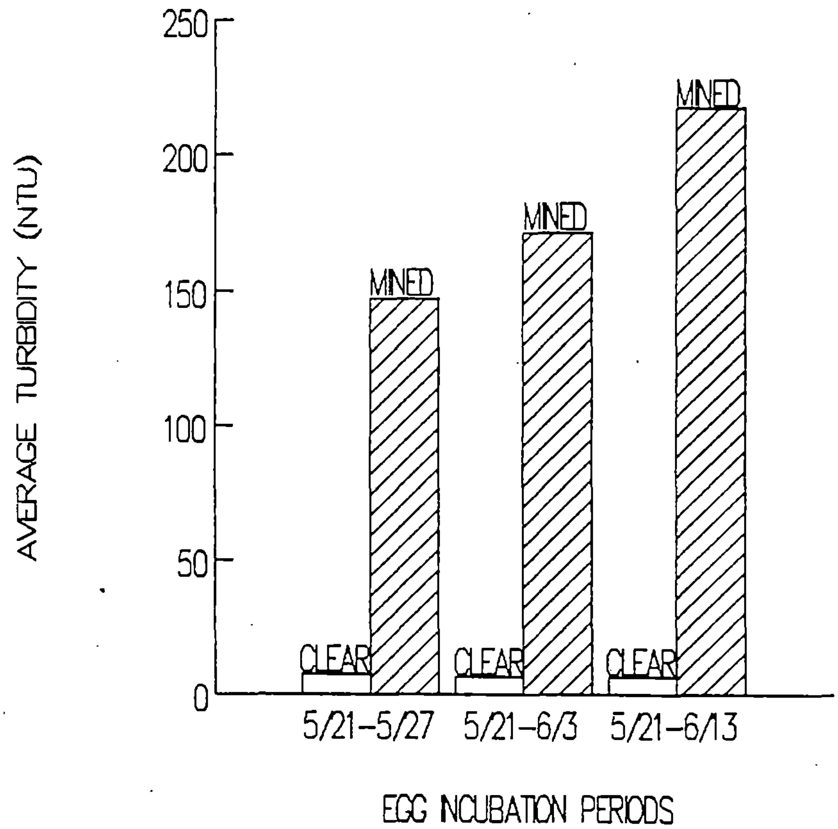

Figure 4. Average turbidity in mined and unmined streams during the incubation periods of three sets of eggs.

Table 4. Total suspended solids levels measured in study sites. N is the total number of measurements taken during the study.

STREAM	Average May 21-May 30	Average May 21-June 3	Average May 21-June 13	Min.	Max.	N
ALBERT	35	23	84	2	290	4
BEDROC	36	44	30	0	66	5
BOLDER	41	41	41	32	51	2
FISH	54	40	40	11	71	3
MCMANS	44	31	23	0	78	4
Average,all unmined sites	42	36	44			
BIRCH at Fish	711	1108	1108	31	2301	4
CROOKED at Boulder	1070	1259	1259	287	1852	3
CROOKED at BLM	290	377	263	63	550	5
CROOKED at Bedrock	345	393	392	174	548	5
DEADWOOD	544	433	350	20	856	5
FAITH	838	580	441	24	857	4
KETCHM	270	248	463	129	1654	6
MAMMOTH	959	959	858	232	1740	5
PEDRO	294	284	274	104	776	15
GILMORE	23	23	21	14	39	4
GOLDSTREAM	130	147	123	31	306	17
Average, all mined sites	498	528	505			

example, TSS measured in Ketchem Creek on June 12 was 1654 mg/L, also about 3.5 times higher than the average level for that creek.

The average TSS concentrations over each of the three egg incubation periods were calculated to determine the average conditions to which the eggs were exposed (Figure 5). For each of the three incubation periods, Arctic grayling eggs were exposed to significantly lower levels of TSS in unmined sites than in mined sites ($p = 0.001$). The average TSS concentrations in unmined sites were 42 mg/L during incubation period 1, 36 during period 2, and 44 during incubation period 3. In mined streams, the average TSS concentrations were 628 mg/L during incubation period one, 670 during period two, and 642 during period three (Figure 5).

The highest TSS levels were found in Birch Creek near Fish Creek (an average of 1108 mg/L) and Crooked Creek at Boulder Creek (an average of 1259 mg/L). Birch Creek at Fish Creek is below Eagle Creek and Golddust Creek; both were intensively mined streams. Placer mining also occurred in Birch Creek itself above the sampling site. TSS levels measured upstream in Crooked Creek near the confluence of Bedrock Creek were lower than downstream at the confluence of Boulder Creek, and averaged 393 mg/L. Three mining sites were located on Crooked Creek between the Bedrock Creek site and the Boulder Creek site.

Settleable solids levels were below the first graduation, 0.5 mL/L, in all of the unmined sites during all of the times measured (Table 5). Of the 21 water samples collected for settleable solids in unmined sites, 19% were recorded as $\leq$ 0.4 mL/L, 76% had trace amounts of sediment, and 5% had no detectable sediment.

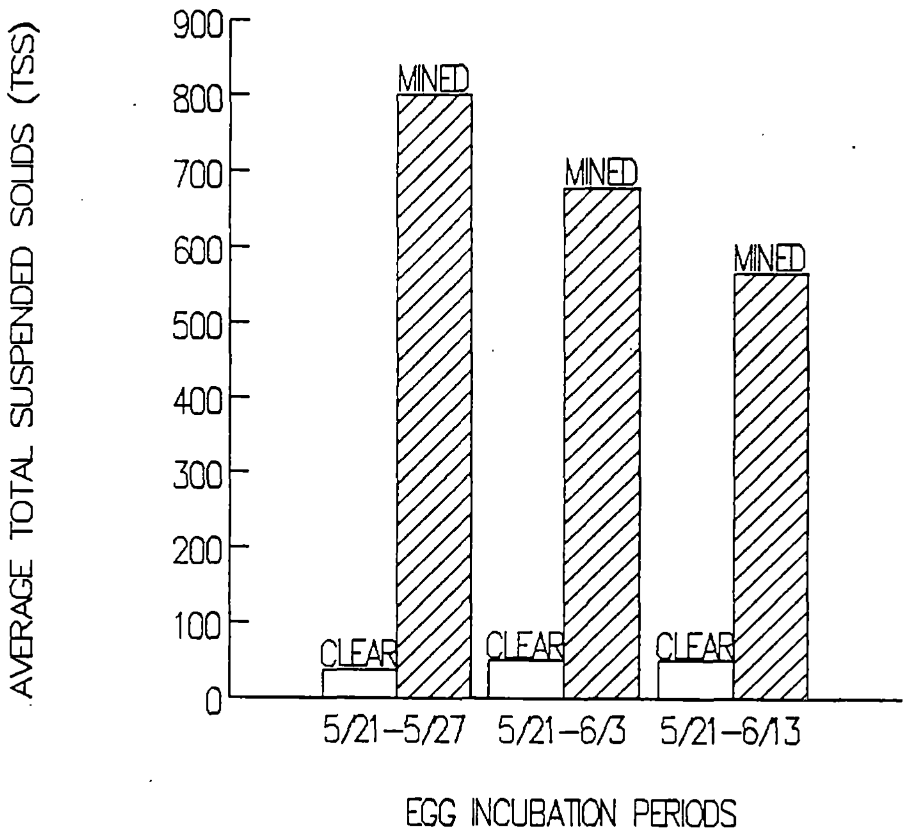

Figure 5. Average total suspended solids in mined and unmined streams during the incubation periods of three sets of eggs.

Table 5. Settleable Solids as measured with an Imhoff cone. Trace (T) indicates not enough sediment to cover the bottom of the cone

SITE	DATE SAMPLED May 21	May 24	May 30	June 3	June 8	June 12
UNMINED SITES						
ALBERT	T	T	-9	T	T	<0.4
BEDROCK	<0.4	T	T	T	T	T
BOULDER	-9	T	-9	<0.4	-9	
FISH	T	<0.4	T	-9	-9	T
MCMANUS	T	T	T	0	-9	T
MINED SITES						
BIRCH AT FISH	1.2	0.8	2.0	1.6	-9	-9
CROOKED AT BOULDER	0.4	<0.4	-9	T	<0.4	-9
CROOKED AT BLM	0.4	<0.4	-9	-9	<0.4	3.2
CROOKED AT BEDROCK	<0.4	0.4	<0.4	<0.4	<0.4	<0.4
DEADWOOD	0.7	0.4	<0.4	T	<0.4	-9
FAITH	1.6	0.7	<0.4	T	-9	T
KETCHUM	<0.4	<0.4	<0.4	<0.4	<0.4	0.6
MAMMOTH	2.0	1.0	-9	-9	<0.4	-9
PEDRO	0.2	-9	0.2	T	-9	-9
GILMORE	T	-9	T	T	-9	-9
GOLDSTREAM	0.4	-9	T	<0.2	-9	-9

Settleable solids levels were higher in mined streams, where they ranged from trace to 3.25 mL/L. About 29% of the 37 water samples collected from mined streams contained settleable solids levels that exceeded 0.5 mL/L. About 60% of the water samples had settleable solids levels that were just below 0.5 mL/L, and the remaining 11% of the samples contained trace amounts of settleable solids.

The pH of unmined streams (mean 6.19, SD = 0.95) was significantly ($p = 0.011$) lower than the pH of mined streams (mean 7.04, SD = 0.48), however the pH of one unmined stream, Bedrock Creek, was exceptionally low (mean 5.01). When Bedrock Creek was excluded from the analysis, pH of the unmined streams (mean 6.78, SD = 0.29) was not significantly ($p = 0.08$) lower than the pH of the mined streams.

The water velocity at the experimental sites was not significantly ($p = 0.54$) different between the mined (mean 0.49 m/s, SD = .026 (1.6 ft/s, SD = 0.86)), and the unmined (mean 0.37 m/s, SD = 0.10 (1.2 ft/s, SD = 0.33)) stream sites.

The water temperature (degrees Celsius) at the experimental sites was not significantly ($p = 0.16$) different between the mined (mean 5.5, SD = 3.3), and the unmined (mean 4.1, SD = 3.1) streams.

Heavy Metals Concentrations

The concentrations of total recoverable arsenic, copper, lead, zinc, and gold were significantly higher ($p \leq 0.001$) in mined streams than in unmined streams (Table 6). Only three samples were above the detection limit for silver, two from mined streams and one from an unmined stream.

TABLE 6. Wilcoxon's ranked sum W test for testing the difference in metals concentrations in samples from mined and unmined streams.

METAL	TYPE	N	MEDIAN	P-VALUE	SIGNIFICANCE
ARSENIC	unmined	10	0.003		
	mined	19	0.027	0.0000	YES
COPPER	unmined	10	0.013		
	mined	19	0.029	0.0007	YES
LEAD	unmined	10	0.001		
	mined	19	0.011	0.0000	YES
ZINC	unmined	10	0.036		
	mined	19	0.077	0.0009	YES
GOLD	unmined	10	<0.006		
	mined	19	0.002	0.0000	YES
SILVER	unmined	10	<0.005		
	mined	19	<0.005	0.8964	NO

None of the unmined streams had total recoverable arsenic concentrations above 0.01 mg/L, while all of the mined streams had total recoverable arsenic concentrations higher than this concentration. Additionally, two of the mined streams had total recoverable arsenic concentrations exceeding the 0.05 mg/L Alaska drinking water standard (18 AAC 80.050) for finished drinking water.

For total recoverable copper, thirty percent of the unmined streams exceeded the Criterion Maximum Concentration (CMC), and 50% of them exceeded the Criterion Continuous Concentration (CCC). In comparison, 84% of the mined streams exceeded the CMC, while 95% of the mined streams exceeded the CCC. The highest concentration of total recoverable copper found in a mined stream was over six times the CCC.

None of the unmined streams exceeded the CCC for total recoverable lead. Eighty-four percent of the mined streams exceeded the CCC for total recoverable lead, with one stream exceeding it by over 2,000%.

Accuracy of the measurements of total recoverable zinc relative to the criterion value was questionable; however, relative to the unmined streams, the mined streams had significantly higher ($p \leq 0.001$) concentrations of total recoverable zinc.

Detectable amounts of total recoverable gold were found in only two of the unmined sites, while detectable amounts of total recoverable gold were found in 18 of the 19 mined sites. There are no water quality criteria for total recoverable gold.

The CMC for total recoverable silver (0.0041 mg/L) was below the detectable limit (0.005 mg/L) of the analytical procedure used. Silver was detected in one unmined stream (0.006 mg/L) and in 2 mined streams (0.016 and 0.008 mg/L). The average of the two samples from the mined streams (0.012 mg/L) contained twice as much total recoverable silver as the sample from the unmined stream (0.006 mg/L). No silver was detected in any other streams.

Toxicity Tests

High current velocities associated with spring breakup caused greater stress on the eggs than was anticipated. Therefore, egg bags were removed from the study streams at shorter intervals than initially planned. The first set was removed after a 10 day period, the second set after four more days, the third set was removed approximately 9 days later. The total incubation period averaged 23 days.

There was no significant linear association between turbidity or total suspended solids and egg survival during any incubation period using a level of significance of 95 percent (Figures 6 - 11). The probability of the slope of the regression line being equal to zero (no linear relationship) for these relationships ranged from $p \geq 0.1$ to $p \geq 0.4$.

Initially eggs appeared to die faster in streams with higher turbidity; however, the distinction between living and dead eggs was less clear after only 10 days than it was after 13 and 23 days. The distinction between living and dead eggs became more clear because the dead eggs became further decomposed, the unhatched, (or hatched) larval fish became visible in living eggs, and also I became more proficient at examining the eggs. Therefore, although Figures 6 and 9 show a negative correlation between egg survival and sedimentation (turbidity and TSS respectively); Figures 7 and 8, and 10 and 11, which reflect more accurate data taken after longer incubation

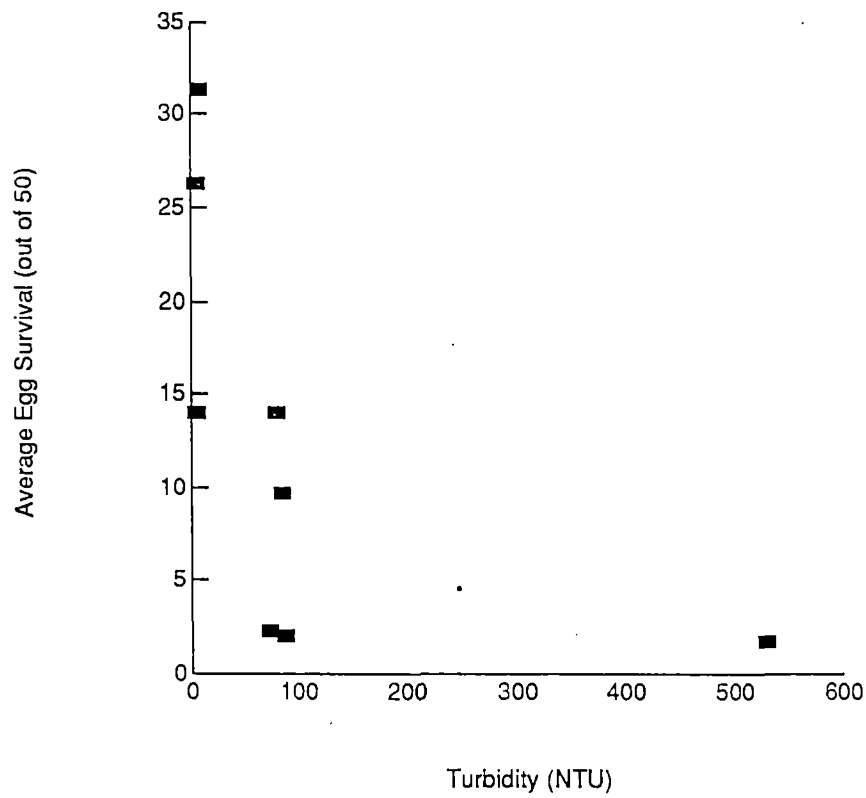

Figure 6: Relationship between average egg survival and average turbidity during the first incubation period, from 5/21 to 5/30, $r^2 = 0.21$.

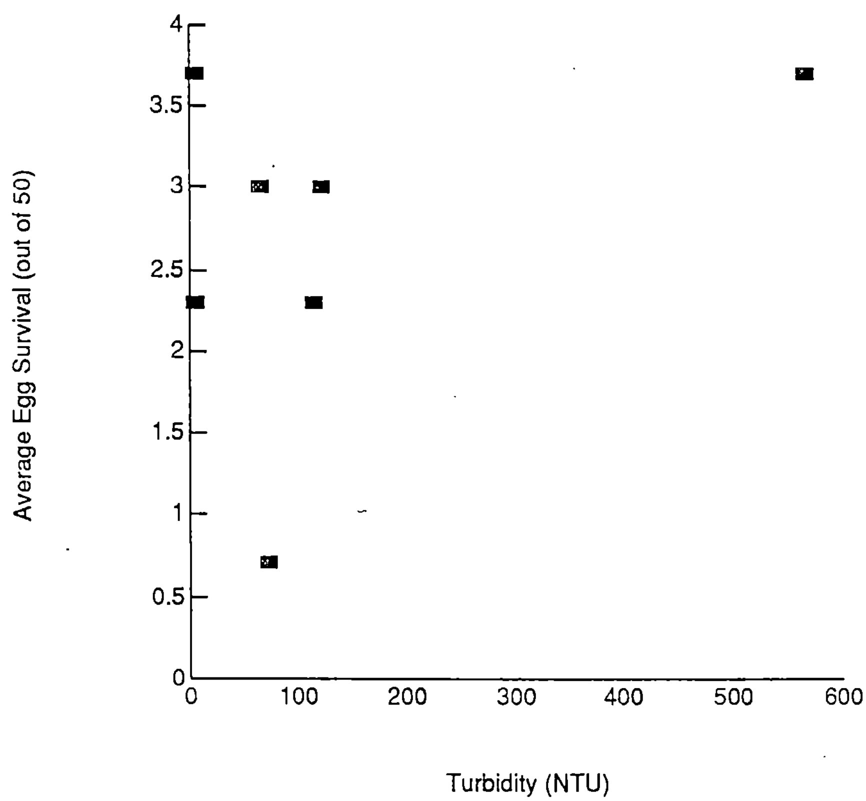

Figure 7: Relationship between average egg survival and average turbidity during the first incubation period, from 5/21 to 6/03, $r^2 = 0.14$.

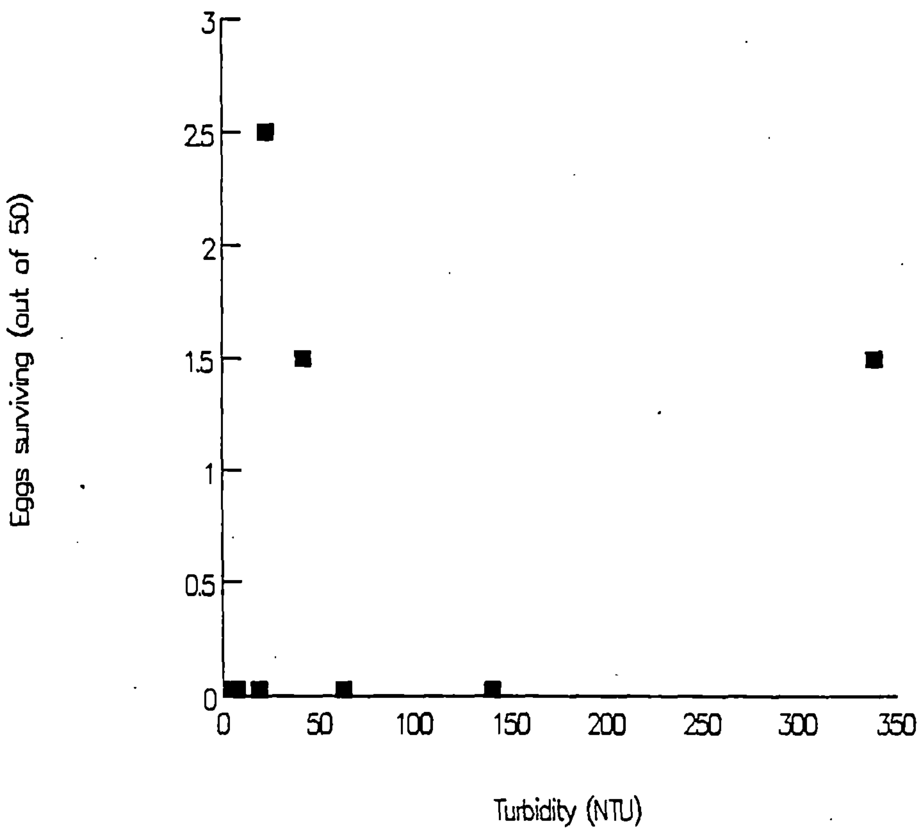

Figure 8. **Relationship between egg survival and turbidity aduring the third incubation period, from 5/21 to 6/13, r^2 = 0.04.**

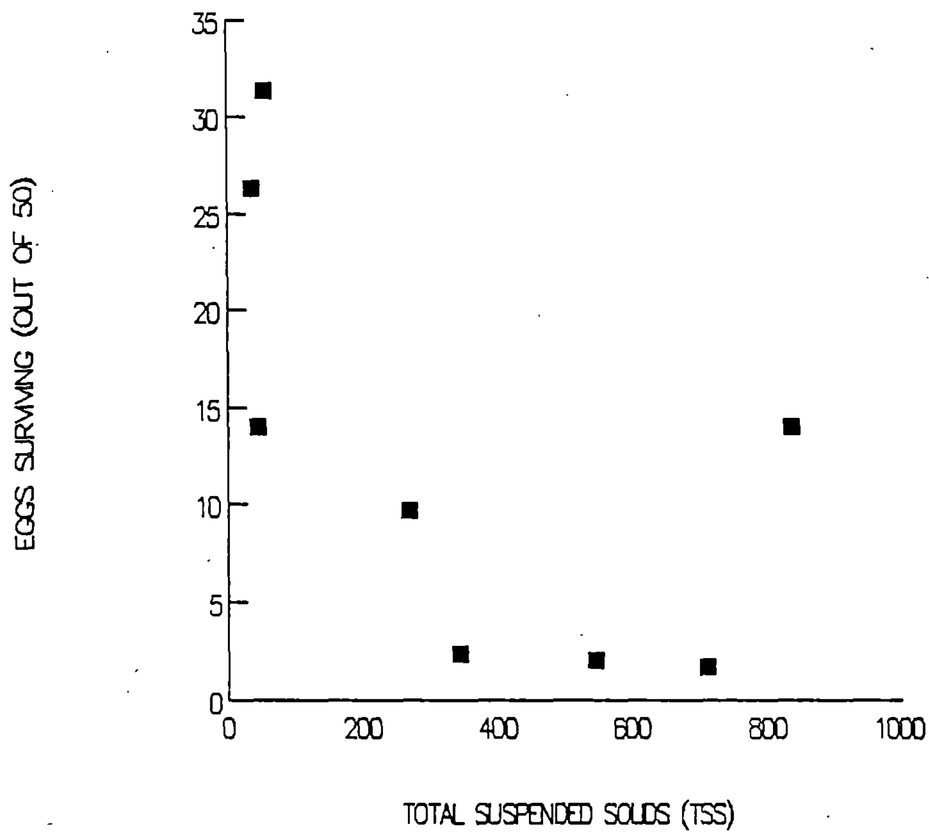

Figure 9. Relationship between egg survival and total suspended solids during the first incubation period, from 5/21 to 5/30, $r^2 = 0.30$.

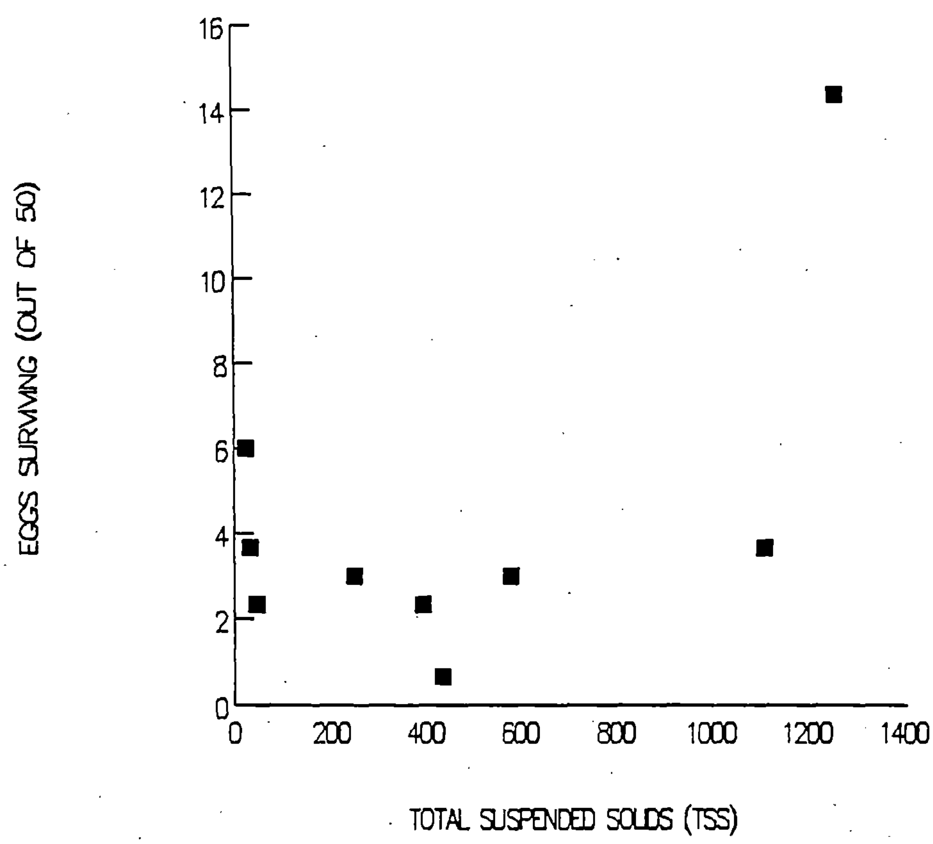

Figure 10. Relationship between egg survival and total suspended solids during the second incubation period, from 5/21 to 6/3, $r^2 = 0.002$.

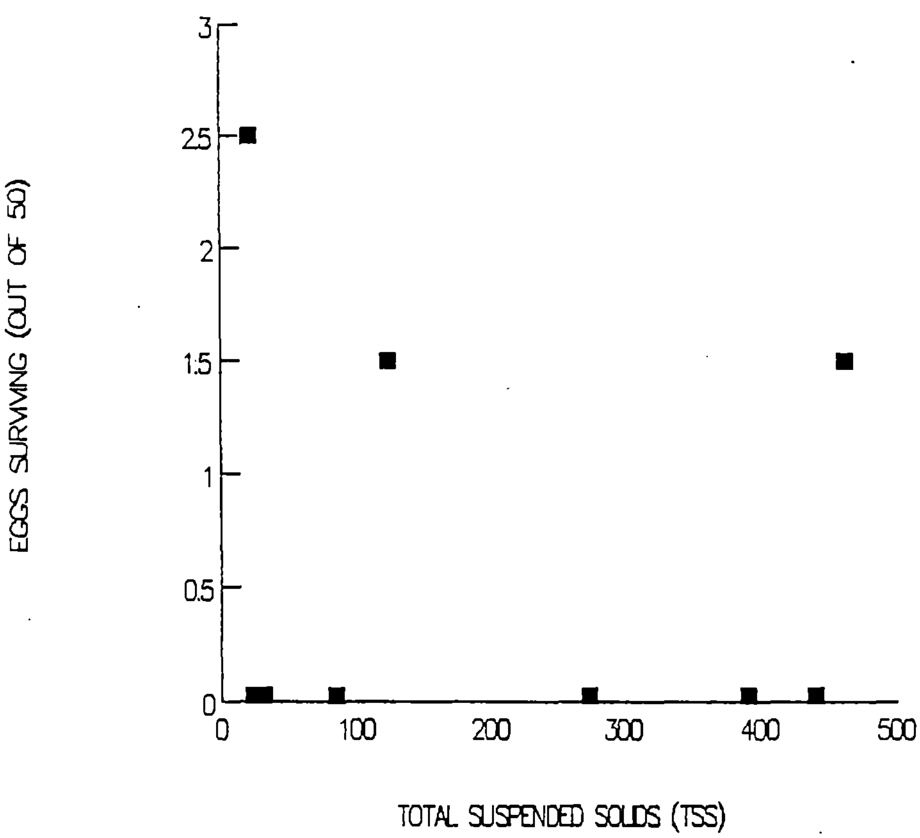

Figure 11. Relationship between egg survival and total suspended solids during the third incubation period, from 5/21 to 6/13, $r^2 = 0.001$.

periods, do not support that correlation. Also, the only eggs that hatched successfully did so in turbid streams.

Discussion

The results of this study did not allow me to reject the null hypothesis ($p = 0.703$) that Arctic grayling eggs will exhibit the same rates of hatching success in clear and turbid streams. It was not shown that placer mining sediment has a significant effect on the survival of Arctic grayling eggs under the conditions of this test. The Arctic grayling eggs survived equally well and in some cases seemingly better in sedimented water as compared to clear water. This does not indicate that placer mining improves the spawning success of Arctic grayling. Many other parts of the spawning act, such as social interaction and burying of the eggs, are quite likely hampered by characteristics of a stream impacted by placer mining pollution such as high turbidity and cemented substrate.

This study showed that, unlike the sac-fry life stage which is highly sensitive to placer mining pollution, the egg development life stage of Arctic grayling was not highly sensitive to placer mining pollution.

BEHAVIORAL STUDIES

In this part of the study I examined the effect of turbidity, a primary characteristic of placer mining effluent, on Arctic grayling avoidance and reactive distance behaviors.

INTRODUCTION

Streams that historically contain abundant Arctic grayling have been found to be nearly devoid of them during prolonged periods of high turbidity such as that caused by placer mining activity (Simmons 1984, Reynolds et al. 1988). It is not known whether the turbid conditions cause the Arctic grayling to die or to leave the turbid stream; however, the recovery of Arctic grayling populations within several years after the return of relatively clear water conditions, in streams that suffered prolonged periods of turbid conditions, suggests that Arctic grayling may avoid a stream when it is highly turbid and return to it when its water conditions improve (J. Hallberg, Fisheries Biologist, Alaska Department of Fish and Game, Fairbanks, pers. com. 1987)

GRAYLING AVOIDANCE OF TURBID WATER

The objectives of this component of the study were to determine if Arctic grayling exhibit a preference for clear or turbid water. Given that turbid waters can damage an Arctic grayling's gills and reduce its feeding ability (Simmons 1984), the fish may demonstrate avoidance of turbid conditions as an adaptive behavior.

To experimentally determine whether Arctic grayling avoid turbid water, I conducted laboratory tests on the preference/avoidance response of Arctic grayling in a chamber that provided a choice between clear water and water of various levels of turbidity. The null hypothesis of this experiment was that Arctic grayling show no preference for either clear or turbid water.

Methods

Juvenile Arctic grayling, (approximately 80 mm long) supplied by Clear Hatchery in plastic bags with oxygen and transported with ice in a cooler, and maintained in a cold water aquarium in a university laboratory, were placed in a plexiglass trough 79 mm deep by 94 mm wide by 952 mm long. Turbid water was pumped into one end, while clear water was pumped into the other; outflow water was syphoned from the center of the trough. This resulted in a trough with one half clear water and one half turbid water with very little mixing at the interface. The Arctic grayling were free to swim between the clear water on one side of the tank and the turbid water on the other side.

The water source for both sides of the choice chamber was chilled laboratory tap water in which the Arctic grayling were held prior to testing. Bentonite clay was used as a source of turbidity.

Turbidity was generated by suspending bentonite clay in the water. Samples of the water, taken over a range of turbidity levels, were analyzed for turbidity (NTU), with a Hach(R) model 16800 portalab nephelometric turbidimeter; total suspended solids (mg/L), using procedures from Standard Methods (APHA et al. 1980); and metals (mg/L), using EPA method number 4.1.4 (total recoverable metals). Except for the addition of bentonite clay, the water on each side of the chamber was treated identically.

In each trial two fish were placed in the choice chamber tank. One was placed on the clear and one on the turbid side. After a 15 minute familiarization period the fish were monitored continuously for 5 consecutive 5-minute periods. The amount of time that they spent on the clear side was recorded. The water condition in which a fish spent the majority of its time was defined as the preferred side for that fish. Seventy-four individual trials were conducted. The turbid side and the clear side were switched during the experiment to check for bias. No bias was observed.

Results

A simple linear regression of TSS (mg/L) against turbidity yielded the formula:

$$TSS = 0.04 + 0.0023\ T$$

where T = turbidity (NTU).

The correlation coefficient (r^2) of this formula of 0.99 indicated a strong relationship between turbidity and total suspended solids (Figure 12). Therefore turbidity is a good indicator of the sediment pollution associated with TSS (mg/L) for this study.

One water sample from the turbid side and one water sample from the clear side was analyzed for metals concentrations by Northern Testing Laboratories, Inc. (NTL). The results of the analysis, in mg/L, are:

Side	arsenic	copper	lead	zinc	gold	silver
Clear	0.022	0.075	0.015	0.076	0.0062	<0.005
Turbid	0.013	0.210	0.002	0.012	0.0036	<0.005.

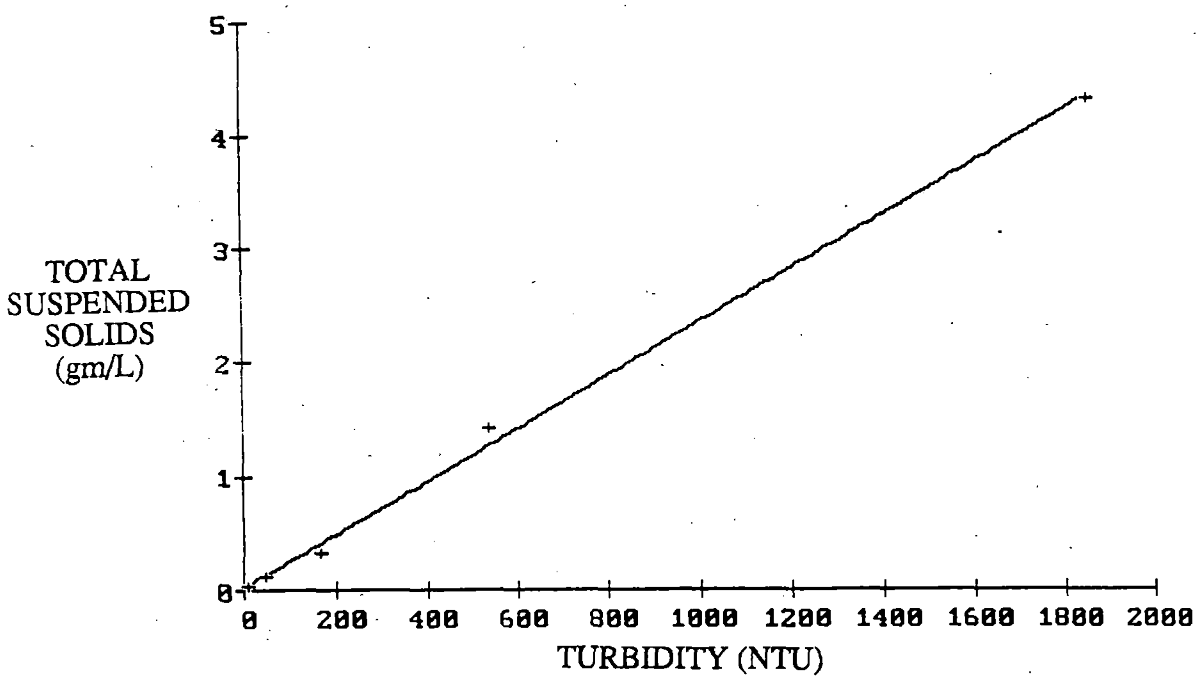

Figure 12. Relationship between total suspended solids and turbidity for suspensions of bentonite clay, $r^2 = 0.99$. The fitted line follows the formula: TSS = 0.04 + 0.0023 T, where T = turbidity (NTU).

There were differences in the metals concentrations between the clear and the turbid sides, but small sample size precludes statistical tests of the significance of these differences. Evaluation of the contribution that these differences made toward Arctic grayling preference of one side over the other is complicated by the acid digestion process used in the metal analysis that may liberate and measure adsorbed metals which were imperceptible to the Arctic grayling during the test.

A series of 74 time trials showed a strong preference of Arctic grayling for clear water. When the turbidity of both sides of the trough was less than 2 NTU the fish did not show a significant preference for either side of the tank. The fish showed increasing preference for the clear side with increasing turbidity of the turbid side (Figure 13).

When the turbid side was 0 to 20 NTU, 38% of the fish preferred the clear side; when 20-40 NTU, 88%; 40 to 60 NTU, 78%; 60 to 80 NTU, 90% and 80 to 100 NTU, 80%. In other words, when the turbid side was 20 NTU or greater, at least 78% preferred the clear side.

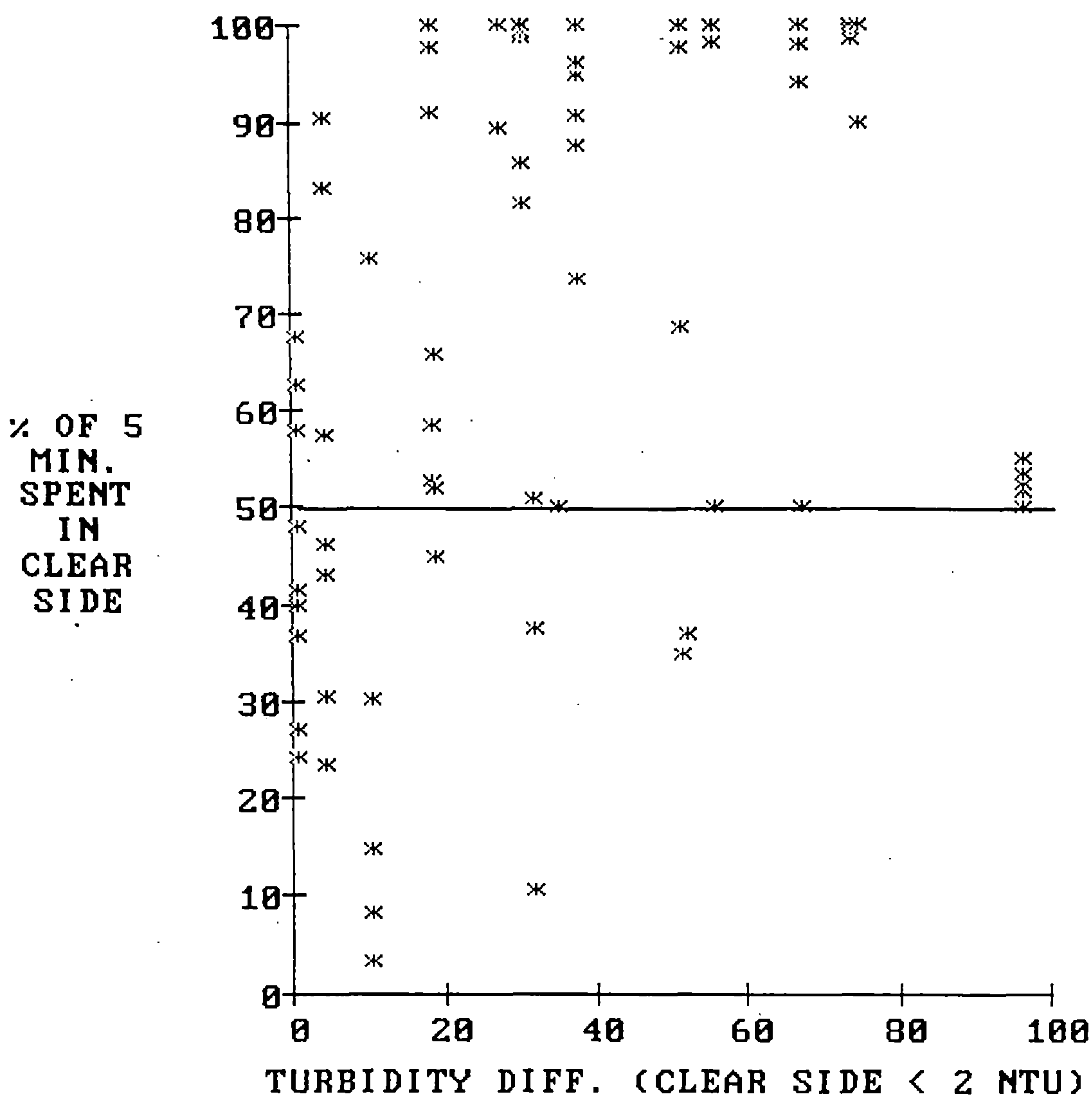

Figure 13. Relationship between the percent of the time that each Arctic grayling spent in the clear side and the turbidity of the turbid side of an avoidance chamber, $r^2 = 0.35$, $p < 0.002$. The regression formula is, Percent time spent in clear side = 55.48 + 0.358 T where T = turbidity (NTU).

Discussion

When given a choice between clear and turbid water, Arctic grayling showed an aversion to turbid water above 20 NTU. Their aversion increased with the turbidity of the water. When Arctic grayling were first exposed to turbid water they often flared their gills and acted disturbed. Damage to Arctic grayling gill lamellae by turbid water has been demonstrated by Simmons (1984). Irritation to their gills may induce Arctic grayling to avoid turbid water. Loss of visual acuity may also account for the avoidance response. My findings of Arctic grayling avoidance of turbid water were probably conservative because the Arctic grayling appeared to spend time in the turbid side during initial tank exploration, and when seeking cover from the more exposed environment of the clear side.

The clear plexiglass choice chamber was an exposed environment, and some Arctic grayling chose to initially move into the most turbid part of the chamber. However, when I covered both sides of the chamber, usually the fish soon moved to the clear water.

The null hypothesis that Arctic grayling would show no preference for either clear or turbid water was rejected because as shown in Figure 13, the fish spent more time in the clear water when the turbidity of the turbid side was increased. Although this laboratory study cannot be extrapolated to suggest that Arctic grayling would necessarily leave a stream as soon as a mining operation muddied the waters, Arctic grayling have not been found to reside in extensively mined drainages such as Birch Creek (Weber and Post 1985, Dames and Moore 1986).

EFFECTS OF TURBIDITY ON REACTIVE DISTANCE OF ARCTIC GRAYLING

This component of the study tested the effects of different turbidity levels on the ability of an Arctic grayling to detect and react to a prey item. The purpose of reactive distance tests were to determine if, or to what degree, feeding behavior is affected by placer mine pollution. The null hypothesis of this experiment is that maximum reactive distance does not vary with turbidity.

The amount of food available to fish depends upon both the abundance of prey items and the ability of the fish to locate and successfully capture the prey. In a naturally clear system, a fish will locate a prey, initiate a pursuit response, and (either successfully or unsuccessfully) capture the prey. The maximum distance at which a predator can locate a specific prey is termed the reactive distance (O'Brien 1979). Reactive distance is not necessarily constant throughout a fish's life. Some fish exhibit increased visual acuity as they grow. This led Dunbrack and Dill (1983) to state that reactive distance is a function of fish length and food width.

Measuring the reactive distance of fish is a way to measure a fish's reduced visual response associated with increased turbidity in a system. In this way reactive distance can be used to measure the impact of turbidity on fish feeding ability.

Increased turbidity reduces the reactive distance of Arctic grayling directly by obscuring the prey, and indirectly by reducing the depth to which light can penetrate in the water and be available to illuminate the prey. Schmidt and O'Brien (1982) found that Arctic grayling reactive distance decreased at light intensities below 20,000 lx. Schmidt and O'Brien suggested that Arctic grayling may be adapted for feeding under the continuous daylight of the Arctic summer, and compared to several sunfish, Arctic grayling are poorly adapted to feeding in low light levels. Although my study does not distinguish between direct and indirect effects of turbidity on Arctic grayling, both are probably significant.

Reactive distance is a one dimensional measurement of the maximum distance at which a predator can locate a prey. However, fish live in a three dimensional world, and they have a potential for locating prey in any of the three dimensions. Therefore, the maximum feeding range of a fish, or the three dimensional volume in which it is capable of locating prey, approximates the cube of its reactive distance (O'Brien et al. 1976, Hairston et al. 1982). This relationship is important because a small decrease in reactive distance results in a large decrease in feeding range.

For example, if a fish's reactive distance is cut in half, there will be an 87.5% reduction in the fish's feeding range. From the fish's standpoint, fewer prey are likely to be found, and there is a greater probability that the prey found will be able to successfully escape (Moore and Moore 1976).

Gardner (1981) states that, "energy intake and thus production of fish populations will be reduced by high (>50 NTU) levels of turbidity." Given constant food delivery rate, energy intake will be reduced by high levels of turbidity because, for many fish, energy intake is dependent on reactive distance and reactive distance is dependent on turbidity. These ideas are supported by studies of Vinyard and O'Brien (1976) that show that the accessibility of plankton prey to visually feeding fish is a function of the reactive distance (the greatest distance at which a fish can locate the prey) of the fish to a particular prey, and by Ringler's (1979) report which states that, "increasing turbidity under laboratory conditions from 2-3 JTU (Jackson Turbidity Units) to 85-90 JTU cut the reaction distance of flounders in half, and doubled the time required in prey capture. For highly mobile decapods the fraction of successful escapes from the flounders increased from 55 percent in clear water to 100 percent in turbid water" (Moore and Moore 1976, cited in Ringler 1979).

As reactive distance decreases, prey encounter rate decreases, therefore feeding rate tends to decrease. Some researchers note this result directly. Feeding by coho *(Oncorhynchus kisutch)* smolts on aquatic insects is inhibited by suspended sediment. Feeding rate decreases to zero above 300 mg/L (Noggle 1978). Hynes (1970) notes that a reduction in water clarity causes fish to stop feeding.

Loss of feeding efficiency is especially important in a northern environment where both feed and feeding time may be extremely limited. While other species of fish may be able to compensate for a reduced feeding efficiency by increasing their foraging time, Arctic grayling forage continuously during the long days of summer and so may have no way to compensate for a reduction in feeding efficiency (Reed 1964).

Many studies show that fish growth is reduced under turbid conditions, without showing a mechanism for the reduction in growth. These studies date from as early as 1956 when Buck reported reduced fish production in turbid ponds in Oklahoma. Exposure to chronic turbidity of varying levels reduces growth in coho salmon and steelhead trout fry (Sigler 1981). Fish raised in clear water achieve several times the weight and length increases of fish raised in water with an average turbidity of 86 NTU (Sigler 1981). More recent studies show that production of juvenile sockeye salmon and returns of adult sockeye salmon are observed to be lower in turbid lake systems than in clear-water lake systems (Lloyd et al. 1987).

In addition to fish production, there are other management considerations that are affected by turbidity. For example, observations by the Alaska Department of Fish and Game (Townsend 1987) indicate that recreational use of streams for sportfishing is reduced in normally clear-water streams when turbidity increases above 8 NTU, and that aerial survey techniques employed in the management of commercial fisheries are hampered at turbidities of 4-8 NTU and above (Lloyd et al. 1987).

As a fish's reactive distance is reduced, the fish not only becomes less able to see its natural prey, but it also becomes less able to see the bait and lures of anglers. This means that if an increase in turbidity from 1 to 25 NTU causes an 87% reduction in feeding range, then it may also cause an 87% reduction in fishing success. Sufficient confidence is placed in this prediction by Lantz (1971), that he indicates that salmon and trout cease feeding once turbidity exceeds 25 JTU by using a drop in angling success as an indicator.

Methods

Juvenile Arctic grayling were placed singly in a 79 mm deep by 94 mm wide by 952 mm long tank that had cooled water slowly flowing through it. The turbidity of the water was raised to various levels with bentonite clay. The turbidity was measured every five minutes during each test with a Hach(R) model 16900 portalab turbidimeter.

Reactive distance tests were conducted with juvenile Arctic grayling about 80 mm long, and also with juvenile Arctic grayling about 135 mm long. After the Arctic grayling was acclimated to the tank, a food item was suspended from the surface tension in an area not visible to the grayling. Wild-caught, dead cladocerans (*Daphnia* sp.) were used for the 80 mm fish, and food pellets measuring about 1x2 mm were used for the 135 mm fish.

Preliminary trials showed that Arctic grayling tended to swim around the tank in a somewhat random manner. When they would notice a daphnid, the Arctic grayling would accelerate, and swim directly towards the daphnid. I measured the distance from the Arctic grayling to the daphnid when the Arctic grayling first reacted.

Results

At any given turbidity level the Arctic grayling reacted to the food item at a distance of from 1 mm to some maximum reactive distance. In other words, sometimes the fish would apparently not notice the food item until it was much nearer to the fish than the fish's maximum reactive distance at that turbidity. Reactive distance at 1 NTU varied from 1 to 11 cm, whereas reactive distance at 95 NTU did not exceed 1 cm. Maximum reactive distance was defined as the farthest 33% of the reactive distances recorded for each turbidity level. Only maximum reactive distances were used in the following analysis.

In the tests with 80 mm fish, 671 feeding trials were conducted and the results of 259 of those trials were accepted as maximum reactive distances. In the tests with 135 mm fish, 109 feeding trials were conducted and the results of 37 of those trials were accepted as maximum reactive distances.

The maximum reactive distance of Arctic grayling decreased with increasing turbidity (Figures 14 and 15). By computing regressions of maximum reactive distance on the natural log of turbidity least squares regression equations were obtained of:

$$MRD_{80} = 6.32 + (-1.06 * \ln(T))$$

where MRD_{80} = maximum reactive distance (cm) of 80 mm fish.

$$MRD_{135} = 14.6 + (-2.79 * \ln(T))$$

where MRD_{135} = maximum reactive distance (cm) of 135 mm fish.

The coefficient of variation (r^2) was 0.63 for the 80 mm fish and 0.86 for the 135 mm fish. The linear association was highly significant ($p < 0.0005$) for the 80 mm fish and ($p < 0.001$) for the 135 mm fish.

Discussion

Reactive volume (RV), the volume of water in which a Arctic grayling feeds, changes with the cube of the reactive distance. Using the regression equation for 80 mm and 135 mm fish given above, at 1 NTU, typical of the turbidity of a clear water stream, the MRD_{80} is 6.32 cm, and the MRD_{135} is 14.6 cm, for juvenile Arctic grayling. At 25 NTU, the Alaska criterion limit for the protection of aquatic life (above a background of near 0 NTU during natural conditions), the MRD_{80} is 2.78 cm and the MRD_{135} 5.62. While this increase in turbidity represents only 55%, and 62% losses of reactive distance, it represents a 91% loss of reactive volume for the 80 mm fish and a 94 % loss of reactive volume for the 135 mm fish.

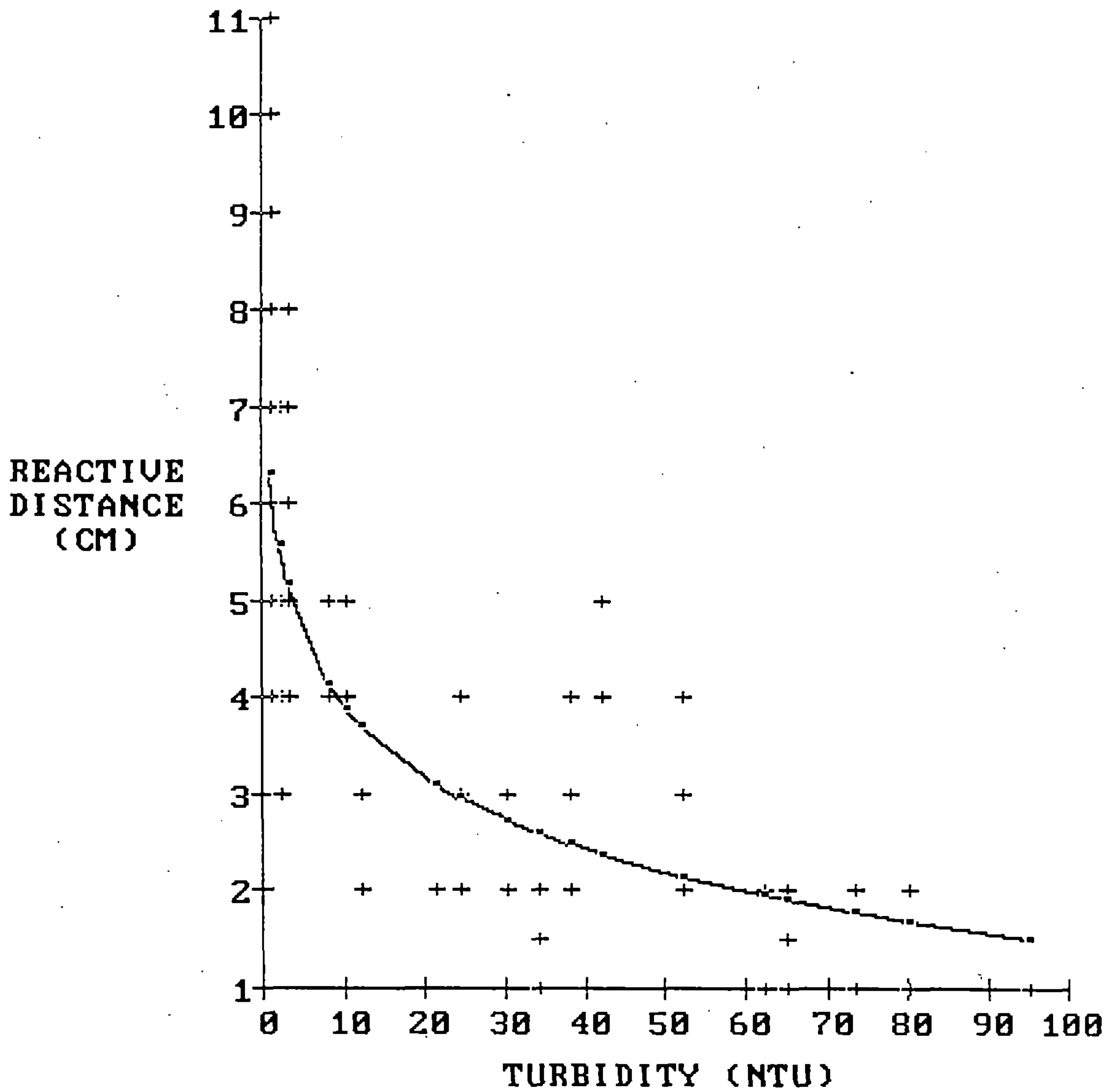

Figure 14. Relationship between reactive distance of 80 mm Arctic grayling and turbidity $r^2 = 0.63$, $p < 0.0005$. The linear regression formula is, $MRD_{80} = 6.32 - 1.06 \ln T$, where MDR_{80} = maximum reactive distance, and T = turbidity (NTU).

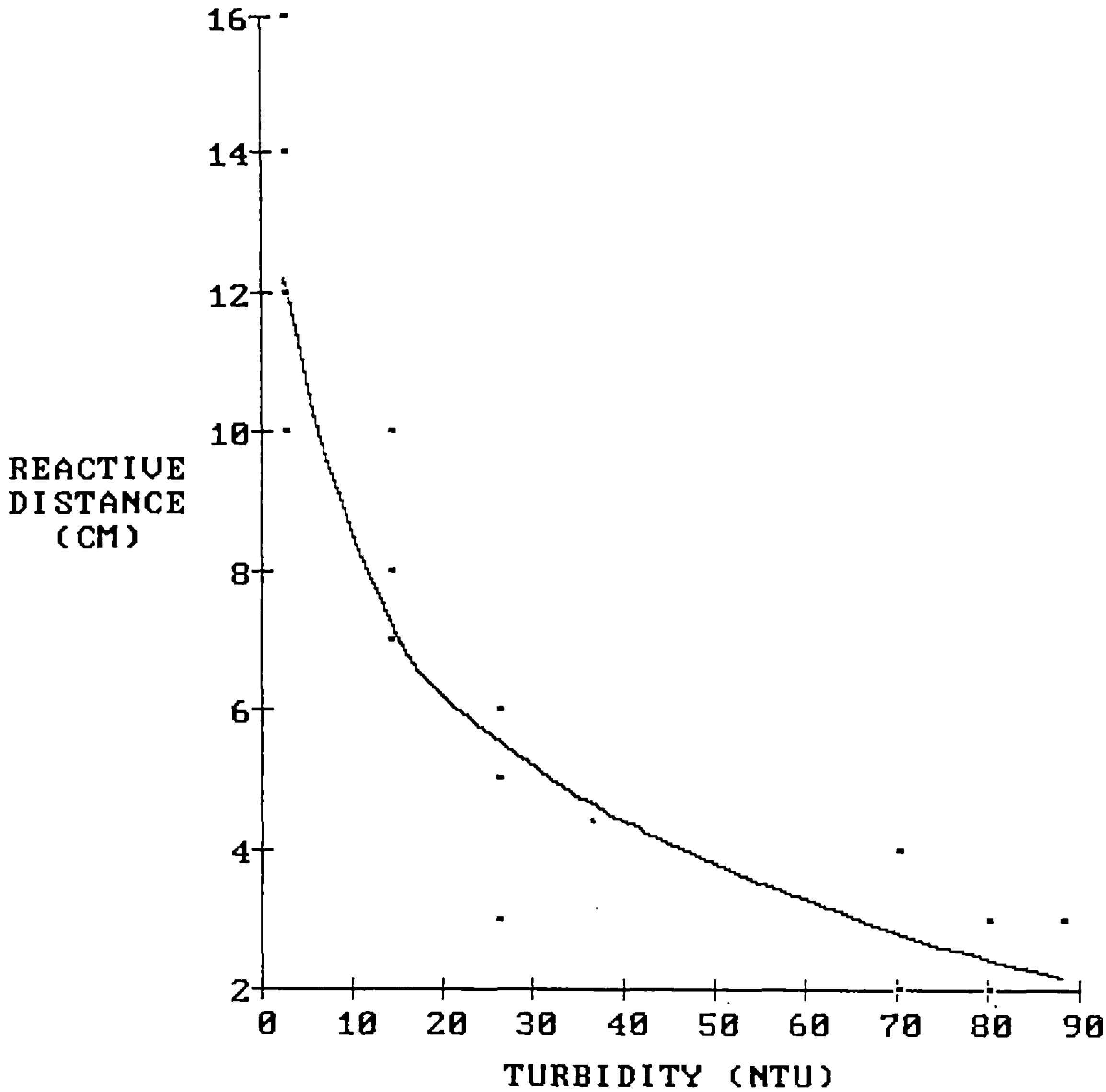

Figure 15. Relationship between reactive distance of 135 mm Arctic grayling and turbidity $r^2 = 0.86$, $p < 0.001$. The regression formula is, $MRD_{135} = 14.6 - 2.79 \ln T$, where MDR_{135} = maximum reactive distance, and T = turbidity (NTU).

The loss of feeding ability due to turbidity related decreased reactive distance may be a critical determinant of Arctic grayling habitat loss because Arctic grayling are sight feeders, and small increases in turbidity greatly reduce the volume of water in which they can feed. Placer mining cam be a particularly important source of turbidity because placer mining often disturbs small headwater streams that are important summer rearing and feeding areas for grayling.

Using the results of this study, I developed a model that may help us understand how the combination of changes in the abundance of stream macroinvertebrates and changes in Arctic grayling reactive volume may effect availability of food to Arctic grayling. Changes in macroinvertebrate density and Arctic grayling reactive volume may be due to placer mining pollution, of which turbidity is an indicator.

Although any model volume would yield similar results, for simplicty grayling reactive volume was assumed to be spherical. It is proportional to reactive distance by the formula:

$$\text{Reactive volume} = (4/3)*\text{pi}*(\text{MRD})^3.$$

The relationship of reactive volume of Arctic grayling with turbidity is shown in Figure 16.

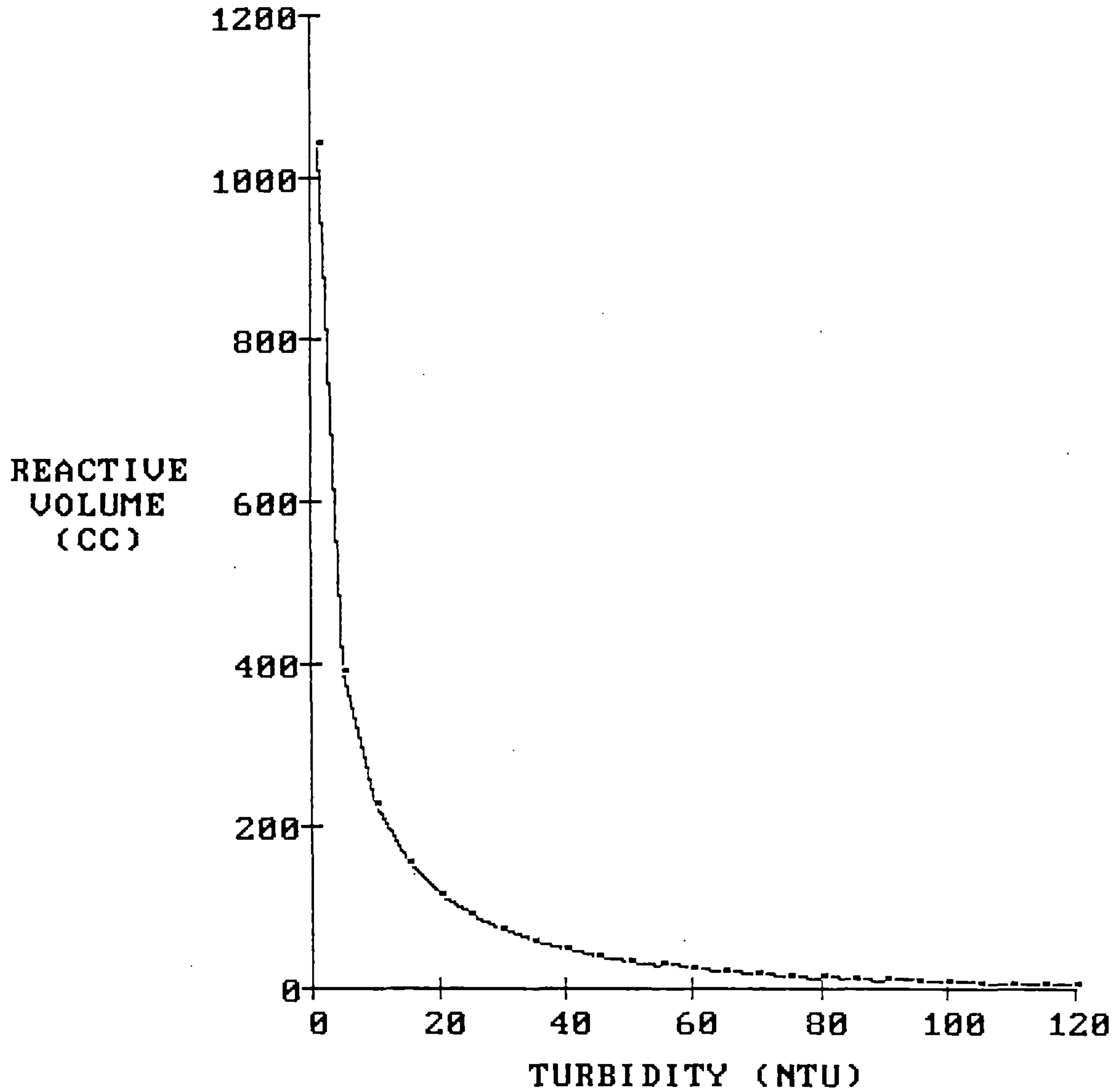

Figure 16. Relationship between reactive volume of Arctic grayling and turbidity. The formula is, $RV = (4/3)*pi*(MRD_{80})^3$, where RV = reactive volume, and MRD_{80} = maximum reactive distance for 80 mm Arctic grayling.

Increased levels of turbidity have also been shown to correlate with decreased density of macroinvertebrates (Wagener 1984). Using Wagener's data I fitted a power regression of macroinvertebrate density (MID), measured as number/0.1m^2, as a function of turbidity, measured in NTU. The following formula obtained the best fit:

$$MID = 97\,T^{-0.38}.$$

The correlation of variation (r^2) of this fit was 0.85. The relationship of macroinvertebrate density and turbidity is shown in Figure 17.

Arctic grayling feed primarily on invertebrate drift (Vascotto 1970). The density of macroinvertebrate drift was assumed to be directly proportional to the density of macroinvertebrates. If water chemistry, velocity, and discharge are held constant (LaPerriere 1983), the abundance of invertebrates available to an Arctic grayling is a function of both the density of drifting macroinvertebrates and the size of the reactive volume of water in which the Arctic grayling feeds.

For example, if an Arctic grayling can see into 100 cm^3 of water, and the macroinvertebrate density is 1 macroinvertebrate per 50 cm^3, then the Arctic grayling is likely to have 2 prey items available at any given time. If 2 prey items is the maximum prey availability, then percent prey availability currently equals 100%. If either reactive volume or macroinvertebrate density is halved, then percent prey availability would be 50%. However, if both were halved, then percent prey availability would be 25%. This is because prey availability equals reactive volume times macroinvertebrate density.

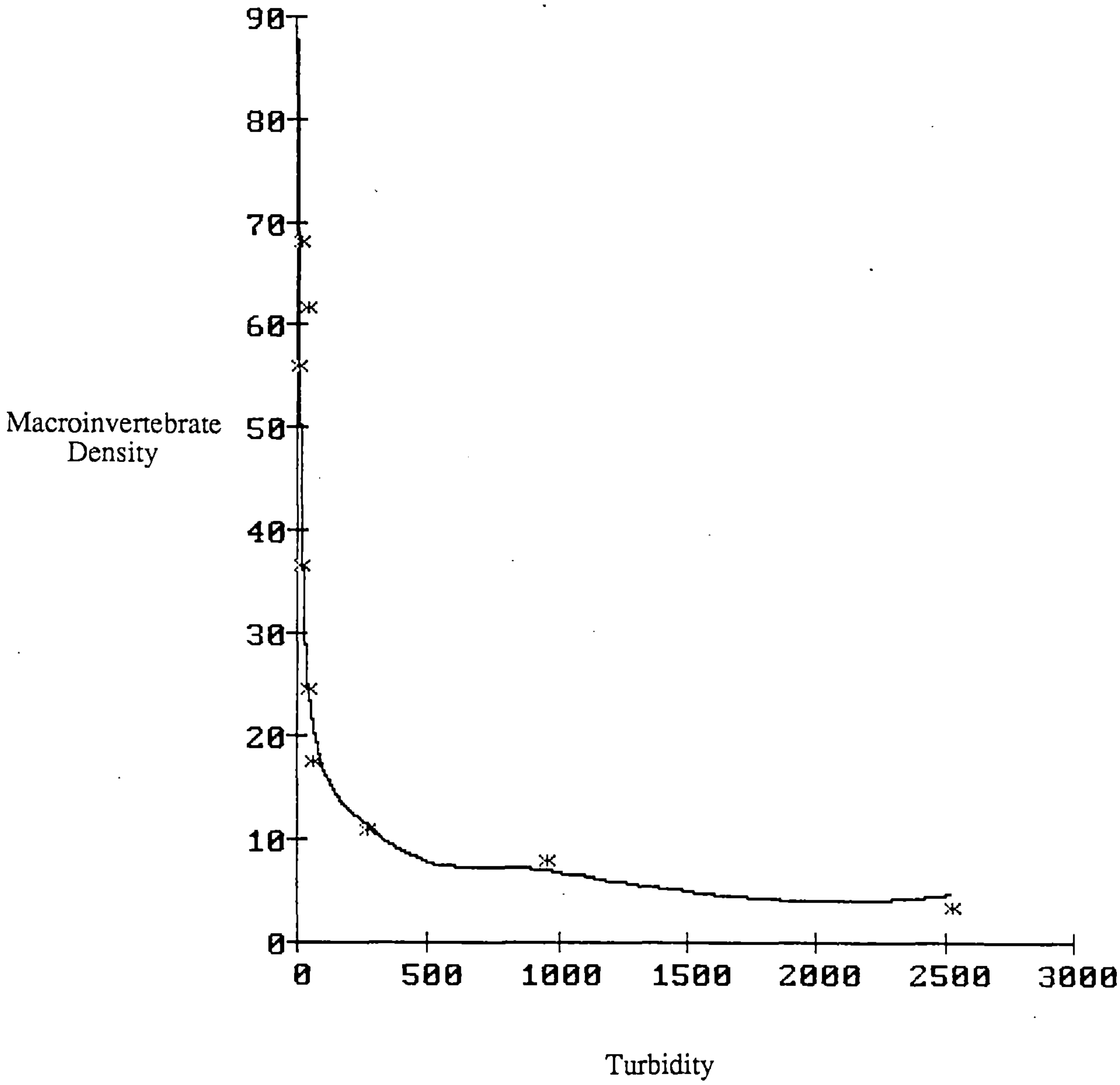

Figure 17. Relationship between invertebrate density nd turbidity $r^2 = 0.85$. The formula is MID = 97 $T^{-0.38}$, where MID = macroinvertebrate density, and T = turbidity (NTU).

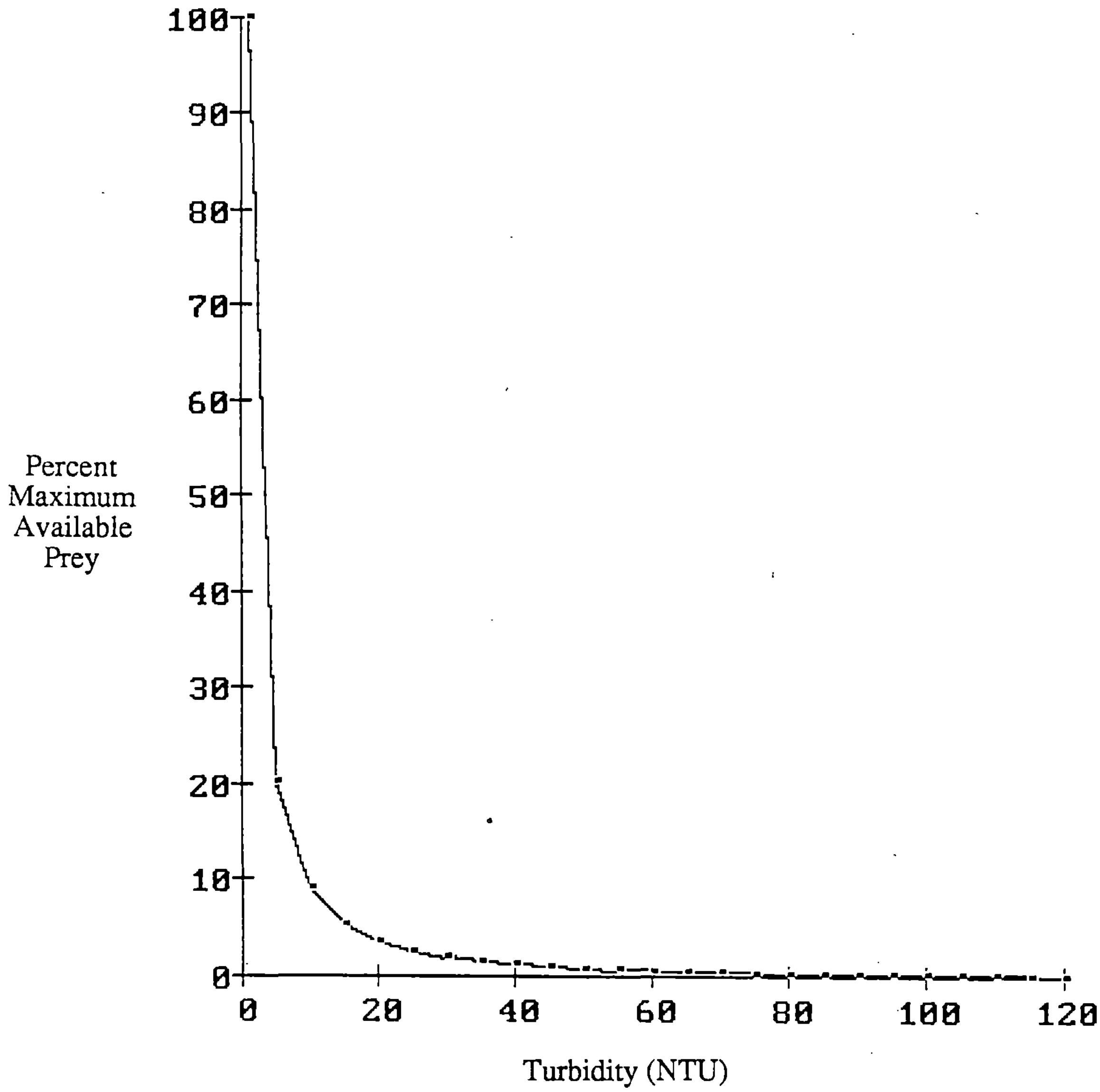

Figure 18. Relationship between percent of maximum available food available to Arctic grayling and turbidity. The formula is, Percent maximum available prey = $(MID/MID_{max})*(RV/RV_{max})$, where MID = macroinvertebrate density, MID_{max} = maximum macroinvertebrate density, RV = reactive volume, and RV_{max} = maximum reactive volume.

Because increased turbidity acts to reduce both macroinvertebrate density and reactive volume, the effect of turbidity on prey availability is multiplicative. The relationship of prey availability and turbidity is shown in Figure 18. The formula for the curve in Figure 17 is:

$$\text{Percent available prey} = (MID/MID_{max})*(RV)/(RV_{max})$$

where MID_{max} = maximum macroinvertebrate density, and RV_{max} = maximum reactive distance.

This model predicts that only 10% of a Arctic grayling's maximum food supply is available to the Arctic grayling at a turbidity of 10 NTU. Assuming that food availability is an important characteristic in Arctic grayling habitat suitability, if we are interested in protecting Arctic grayling habitat, consideration of only invertebrate density or turbidity would be incomplete. To better understand and protect Arctic grayling we must look at resources critical to Arctic grayling from a fish's point of view. Models can aid us in this effort.

CONCLUSIONS

How should the discharge of sediment laden effluents into Arctic grayling habitat be regulated? What extent of sedimentation is compatible with the protection of Arctic grayling and their habitat? Toxicity tests can measure the direct mortality response of individuals of a species. Certain substances or conditions, while not producing a detectable increase in mortality, may alter the habitat in such a way that it becomes less suitable. Two critical aspects of high quality Arctic grayling rearing habitat are the numbers of desirable prey available to the fish and how successful the fish are in finding and capturing these prey. This study examined the effects of increased turbidity, caused by placer mining effluent, on Arctic grayling mortality. This research was extended to examine the effects of elevated turbidity of Arctic grayling habitat on Arctic grayling behavior, as measured by avoidance of the turbid conditions, and the ability of the Arctic grayling to find and capture prey.

The four major life stages of Arctic grayling are the adult, the juvenile, the sac-fry, and the egg stage. Earlier studies addressed the effects of increased turbidity from placer mining on adult and sac-fry Arctic grayling (Simmons 1984, Reynolds et al. 1988). Toxicity studies of adult Arctic grayling did not show mortality due to exposure to turbid conditions for up to 13 days. Sac-fry, tested under similar conditions, exhibited higher mortality rates in turbid water than in clear water after only 24 hours. These earlier studies were important in understanding some of the effects of turbid conditions on Arctic grayling; however, because not all life stages were addressed, the most vulnerable life stage may have been omitted. This study tested the remaining life stages of Arctic grayling: egg and juvenile.

Toxicity tests conducted on the effects of turbidity on Arctic grayling juvenile and egg stages did not show a higher mortality rate in turbid water than in clear water. With the results of this study, I conclude that moderately elevated turbidity levels from placer mining do not result in increased mortality rates of adult, juvenile, or egg stages of Arctic grayling; however, sac-fry do exhibit higher mortality rates when exposed to turbid conditions.

The avoidance of turbid water by Arctic grayling was determined by placing juvenile Arctic grayling in a choice chamber with flowing clear water, and flowing water made turbid with bentonite clay. The fish were watched during a series of five-minute trials, and water in which they spent the majority of their time was designated as the water that they preferred for that trial. When the turbidity of the turbid water was increased, the fish spent more time in the clear water.

Changes in the ability of the grayling to find and capture food were determined by testing the reactive distance of the fish under a range of turbidities. Feeding trials were conducted in flowing water troughs with turbidity held constant throughout the trial. The grayling showed a strong decline in their ability to detect and react to prey items as the turbidity increased.

The loss of reactive distance of Arctic grayling translates into a much larger loss of reactive volume because reactive volume is proportional to reactive distance cubed. With even modest increases in turbidity, such as to 25 NTU, the grayling will have its reactive distance reduced by over 50%, which translates to a reduction of reactive volume of over 90%. At this level, particularly if the periods of elevated turbidities are prolonged with no "relief" periods, the grayling may not be able to find sufficient food to survive. When the loss of effective feeding volume available to Arctic grayling due to increased turbidity is combined with a decline in the numbers of macroinvertabrate food organisms due to sediment pollution (Wagener and LaPerriere 1985, Weber and Post 1985), it becomes apparent that turbidity levels over 25 NTU (the current Alaska water quality standard for the protection of fish and wildlife) cause serious degradation to Arctic grayling habitat. The rapid degradation of Arctic grayling habitat with loss of clarity found in this study support use of a standard of 5 NTU above natural conditions (presently the State of Alaska drinking water supply standard) as an appropriate standard for the protection of Arctic graying habitat.

Tests of reactive distance and avoidance demonstrated that, although certain life stages of Arctic grayling do not die when exposed to placer mining effluents, grayling do exhibit behavioral responses, such as reduced reactive distance in turbid water and avoidance of the turbid water, that may render them less able to survive in turbid water conditions. If Arctic grayling avoid turbid water in streams to the extent that they did in my laboratory tests, then streams made turbid by placer mining effluents represent lost habitat to Arctic grayling, reduced population levels within a drainage, and lost sport fishing opportunities to anglers.

Although the well-being of an individual Arctic grayling in a clinical toxicity test may be isolated from the well-being of Arctic grayling habitat, the well-being of our wild Arctic grayling populations depends on the care we give to the places where they live.

APPENDIX

Table 7. Key to the abbreviations used for site names.

Abbr.	Name	Location
ALBERT	Albert Creek	Below Steese Highway Bridge
BEDROC	Bedrock Creek	Above mouth
BOLDER	Boulder Creek	Above mouth
FISH	Fish Creek	Above mouth
MCMANS	McManus Creek	Above confluence with Faith Creek
BIRFSH	Birch Creek at Fish	Above confluence with Fish Creek
CRKBLD	Crooked Creek at Boulder Creek	Above confluence with Boulder Creek
CRKBLM	Crooked Creek at Central	Above Steese Highway Bridge
CRKBRK	Crooked Creek at Bedrock Creek	Above confluence with Bedrock Creek
DEDWOD	Deadwood Creek	Below Circle Hot Springs Road
FAITH	Faith Creek	Above confluence with McManus Creek
KETCHM	Ketchem Creek	Below Circle Hot Springs Road
MAMOTH	Mammoth Creek	Below Steese Highway Bridge
PEDRO	Pedro Creek	Above mouth
GILMORE	Gilmore Creek	Above mouth
GLDSTM	Goldstream Creek	Above Ballaine Road Bridge

Table 8. Data from Arctic grayling egg toxicity test. Blanks and dashes represent missing values.

Site	Date	Temp. oC	Turb. NTU	SS ml/L	TSS mg/L	Vel. m/s	Egg1	Egg2	Egg3	Avg.
							surviving out of 50			
ALBERT	5/21	0.6	14.0	0.1	-	-				
BEDROC	5/21	0.6	4.0	0.2	31.2	-				
BOLDER	5/21	0.3	11.0	0.1	32.1	-				
BRHFSH	5/21	0.6	250.0	1.2	1015.1	-				
CRKBLD	5/21	1.7	85.0	0.4	287.0	-				
CRKBLM	5/21	1.7	125.0	0.4	298.1	-				
CRKBRK	5/21	2.2	33.0	0.2	313.0	-				
DEDWOD	5/21	0.6	55.0	0.7	855.5	-				
FAITH	5/21	1.1	55.0	1.6	857.2	-				
FISH	5/21	0.6	3.1	0.1	37.3	-				
KETCHM	5/21	0.6	130.0	0.2	374.7	-				
MAMOTH	5/21	0.6	275.0	2.0	1740.1	-				
MCMANS	5/21	0.6	4.5	0.1	-	-				
ALBERT	5/24	5.0	6.5	0.1	34.6	-				
BEDROC	5/24	3.0	5.5	0.1	65.9	-				
BOLDER	5/24	2.0	14.0	0.2	50.5	-				
BRHFSH	5/24	-	390.0	0.8	1086.6	-				
CRKBLD	5/24	-	80.0	0.2	1852.3	-				
CRKBLM	5/24	5.0	130.0	0.2	282.6	3.0				
CRKBRK	5/24	5.0	120.0	0.5	548.3	-				
DEDWOD	5/24	4.0	120.0	0.4	511.9	-				
FAITH	5/24	3.0	120.0	0.7	818.2	-				
FISH	5/24	-	4.5	0.2	-	-				
KETCHM	5/24	-	65.0	0.2	244.1	-				
MAMOTH	5/24	3.0	325.0	1.0	1740.1	-				
MCMANS	5/24	-	6.2	0.1	9.6	-				
BEDROC	5/30	4.5	4.6	0.1	11.4	-	33.0	25.0	21.0	26.3
BIRFSH	5/30	-	950.0	2.0	30.7	-	0.0	3.0	2.0	1.7
CRKBRK	5/30	6.0	61.0	0.2	173.6	-	0.0	3.0	4.0	2.3
DEDWOD	5/30	5.0	84.0	0.2	264.9	-	5.0	1.0	0.0	2.0
FAITH	5/30	-	64.0	0.2	-	-	6.0	21.0	15.0	14.0
FISH	5/30	-	18.0	0.1	71.1	0.8	36.0	37.0	21.0	31.3
KETCHM	5/30	6.0	61.0	0.2	191.3	-	12.0	7.0	10.0	9.7
MCMANS	5/30	-	3.9	0.1	78.4	-	10.0	24.0	8.0	14.0

Table 8, continued.

Site	Date	Temp. oC	Turb. NTU	SS ml/L	TSS mg/L	Vel. m/s	Egg1	Egg2 surviving out of 50	Egg3	Avg.
ALBERT	6/03	8.0	3.5	0.1	11.4	1.6	5.0	3.0	10.0	6.0
BEDROC	6/03	5.0	1.6	0.1	66.3	1.3	5.0	1.0	1.0	2.3
BIRFSH	6/03	5.0	680.0	1.6	2301.4	0.9	0.0	11.0	0.0	3.7
CRKBLD	6/03	6.0	29.0	0.1	1639.1	1.2	3.0	35.0	5.0	14.3
CRKBRK	6/03	7.0	244.0	0.2	539.0	2.0	4.0	1.0	2.0	2.3
DEDWOD	6/03	8.0	32.0	0.1	99.1	1.4	1.0	0.0	1.0	0.7
FAITH	6/03	7.0	21.0	0.1	64.3	0.3	0.0	3.0	6.0	3.0
KETCHM	6/03	9.0	228.0	0.2	182.4	1.4	4.0	4.0	1.0	3.0
MCMANS	6/03	5.0	1.9	0.0	5.3	1.2	3.0	3.0	5.0	3.7
ALBERT	6/08	9.0	2.5	0.1	1.7	-				
BEDROC	6/08	6.0	4.7	0.1	-	-				
BOLDER	6/08	-	-	-	-	-				
CRKBLD	6/08	7.0	-	0.2	-	-				
CRKBLM	6/08	10.0	150.0	0.2	118.6	-				
CRKBRK	6/08	9.0	295.0	0.2	388.4	-				
DEDWOD	6/08	8.0	11.0	0.2	19.8	-				
KETCHM	6/08	8.0	350.0	0.2	129.2	-				
MAMOTH	6/08	12.0	440.0	0.2	232.3	-				
ALBERT	6/12	6.0	-	0.2	290.1	-	0.0	0.0	0.0	0.0
BEDROC	6/12	-	4.8	0.1	7.4	-	0.0	0.0	0.0	0.0
CRKBLM	6/12	7.5	650.0	3.3	63.5	-	-	-	-	-
CRKBRK	6/12	-	87.0	0.2	-	-	0.0	0.0	0.0	0.0
DEDWOD	6/12	7.0	-	-	-	-	-	-	-	-
FAITH	6/12	12.0	55.0	0.1	24.1	-	0.0	0.0	0.0	0.0
KETCHM	6/12	8.0	1200.0	0.6	1653.6	-	1.0	2.0	0.0	1.5
MCMANS	6/12	9.5	4.4	0.1	0.5	-	0.0	0.0	0.0	0.0
GLDSTM	6/12	-	42.0	-	123.0	-	10.0	3.0	0.0	1.5
GILMOR	6/12	-	22.9	-	21.0	-	3.0	4.0	1.0	2.5
PEDRO	6/12	-	18.4	-	274.0	-	1.0	0.0	0.0	0.0

Table 9. Heavy metals and arsenic data taken during the Arctic grayling fingerling toxicity test, 1986.

	Date	ARSENIC mg/L	COPPER mg/L	LEAD mg/L	ZINC mg/L
UNMINED CREEKS					
Twelvemile Creek	7/30	<0.001	0.03	<0.001	0.022
Twelvemile Creek	7/30	<0.001	0.02	<0.001	0.015
Twelvemile Creek	7/30	<0.001	<0.02	<0.001	0.006
Twelvemile Creek	8/1	<0.001	0.02	<0.001	<0.005
Twelvemile Creek	8/1	<0.001	<0.02	<0.001	0.008
Twelvemile Creek	8/1	<0.001	<0.02	<0.001	<0.005
Twelvemile Creek	8/3	0.101	0.16	0.15	0.212
Twelvemile Creek	8/3	0.091	0.16	0.147	0.218
Twelvemile Creek	8/3	<0.001	<0.02	<0.001	0.007
MINED CREEKS					
Birch Creek	7/30	0.007	0.03	0.003	0.036
Birch Creek	7/30	0.007	0.04	0.003	0.068
Birch Creek	7/30	0.009	0.03	<0.001	0.045
Birch Creek	8/1	0.013	0.03	0.008	0.039
Birch Creek	8/1	0.008	0.03	0.005	0.033
Birch Creek	8/1	0.007	0.03	0.004	0.055
Birch Creek	8/3	0.011	0.03	0.001	0.020
Birch Creek	8/3	0.01	0.03	0.004	0.031
Birch Creek	8/3	0.011	0.04	0.006	0.050

Table 10. Heavy metals and arsenic data for the Arctic grayling egg toxicity test, analyzed by Northern Testing Laboratory.

CREEK	DATE	ARSENIC mg/L	COPPER mg/L	LEAD mg/L	ZINC mg/L	GOLD mg/L	SILVER mg/L
MCMANUS	5/23	0.005	0.005	<0.001	0.023	<0.0006	<0.005
	6/23	0.001	0.016	<0.001	0.020	<0.0006	<0.005
BEDROCK	5/23	0.008	0.010	0.002	0.056	0.0007	<0.005
	6/23	0.004	0.024	0.002	0.040	<0.0006	<0.005
ALBERT	5/23	0.002	0.021	<0.001	0.038	<0.0006	0.006
	6/23	0.001	0.018	0.001	0.063	<0.0006	<0.005
FISH	5/23	<0.001	0.010	<0.001	0.034	0.0010	<0.005
	6/23	<0.001	0.010	<0.001	0.020	<0.0006	<0.005
BOULDER	5/23	0.005	0.009	0.001	0.034	<0.0006	<0.005
	6/23	0.004	0.016	0.002	0.046	<0.0006	<0.005
PEDRO	5/23	0.072	0.015	0.034	0.082	0.0014	<0.005
GILMORE	5/23	0.021	0.026	0.011	0.093	0.0031	<0.005
GLDSTRM	5/23	0.021	0.022	0.006	0.082	0.0017	<0.005

Table 10, continued.

CREEK	DATE	ARSENIC mg/L	COPPER mg/L	LEAD mg/L	ZINC mg/L	GOLD mg/L	SILVER mg/L
DEADWD	5/23	0.033	0.029	0.018	0.094	0.0044	<0.005
	6/23	0.015	0.044	0.003	0.060	0.0017	<0.005
FAITH	5/23	0.038	0.035	0.017	0.070	0.0024	<0.005
	6/23	0.006	0.010	0.003	0.025	<0.0006	<0.005
CRKD AT BLDR	5/23	0.011	0.025	0.017	0.077	0.0022	<0.005
	6/23	0.027	0.029	0.009	0.045	0.0026	<0.005
CRKD AT BEDRK	5/23	0.033	0.040	0.012	0.060	0.0035	<0.005
	6/23	0.021	0.017	0.004	0.039	0.0010	<0.005
CRKD AT BLM	5/23	0.019	0.023	0.003	0.028	0.0016	<0.005
	6/23	0.028	0.042	0.027	0.100	0.0034	0.016
KETCHEM	5/23	0.020	0.015	0.043	0.090	0.0024	<0.005
	6/23	0.083	0.080	0.066	0.239	0.0113	<0.005
MAMMOTH	5/23	0.040	0.065	0.018	0.090	0.0059	<0.005
	6/23	0.028	0.029	0.011	0.068	0.0036	<0.005
BIRCH AT FISH	5/23	0.017	0.034	0.011	0.067	0.0021	0.008
	6/23	0.029	0.037	0.011	0.169	0.0016	<0.005

Table 11. Percent of five-minute test period spent in clear side at various turbidity levels. Data for avoidance study.

NTU TURBID SIDE	% TIME CLEAR SIDE	NTU TURBID SIDE	% TIME CLEAR SIDE
0.2	24.5	30.0	98.8
0.2	36.8	30.0	99.3
0.2	41.5	30.0	100.0
0.2	58.0	31.3	10.5
0.2	62.7	31.3	37.5
0.2	67.7	31.3	51.0
0.3	24.2	34.9	50.0
0.3	27.0	37.4	73.7
0.3	40.0	37.4	87.5
0.3	48.2	37.4	90.7
4.0	23.3	37.4	95.0
4.0	30.5	37.4	96.2
4.0	43.2	37.4	100.0
4.0	46.2	51.0	35.0
4.0	57.3	51.0	68.8
4.0	83.0	51.0	97.7
4.0	90.3	51.0	100.0
10.0	3.3	51.0	100.0
10.0	8.3	51.9	37.0
10.0	14.7	55.5	50.2
10.0	30.3	55.5	98.2
10.0	75.7	55.5	100.0
17.9	90.8	67.0	50.0
18.0	52.8	67.0	94.2
18.0	58.5	67.0	98.0
18.0	97.8	67.0	100.0
18.0	100.0	73.5	98.3
18.0	100.0	73.5	98.7
18.3	44.8	73.5	99.5
18.3	51.8	73.5	100.0
18.3	65.8	74.5	90.0
27.2	89.3	74.5	100.0
27.2	100.0	96.0	50.0
27.2	100.0	96.0	51.7
30.0	81.7	96.0	52.5
30.0	85.5	96.0	53.5
30.0	85.7	96.0	55.0

Table 12. Reactive distance (cm) of juvenile Arctic grayling in water of various turbidities (NTU).

Reactive distance cm	turbidity NTU	Reactive distance cm	turbidity NTU	Reactive distance cm	turbidity NTU
11	1	5	1	4	8
11	1	5	1	4	8
11	1	5	1	5	10
10	1	5	1	5	10
10	1	5	1	5	10
10	1	5	1	5	10
10	1	5	1	5	10
10	1	5	1	5	10
9	1	5	1	4	10
9	1	5	1	4	10
9	1	4	1	3	12
9	1	4	1	3	12
9	1	4	1	3	12
8	1	4	1	3	12
8	1	4	1	3	12
8	1	4	1	2	12
8	1	4	1	2	12
8	1	4	1	2	12
7	1	4	1	2	12
7	1	7	2	2	12
7	1	7	2	2	12
7	1	5	2	2	12
7	1	5	2	2	12
7	1	5	2	2	12
7	1	4	2	2	12
7	1	4	2	2	21
6	1	4	2	4	24
6	1	4	2	4	24
6	1	4	2	4	24
6	1	3	2	4	24
6	1	8	3	4	24
6	1	7	3	4	24
6	1	6	3	4	24
5	1	6	3	4	24
5	1	6	3	3	24
5	1	5	3	3	24
5	1	5	3	3	24
5	1	4	3	3	24
5	1	5	8	3	24
5	1	5	8	3	24
5	1	4	8	3	24
5	1	4	8	3	24

Table 12, continued.

Reactive distance cm	turbidity NTU	Reactive distance cm	turbidity NTU	Reactive distance cm	turbidity NTU
5	1	4	8	3	24
5	1	4	8	3	24
5	1	4	8	3	24
3	24	3	38	2	52
3	24	3	38	2	52
3	24	2	38	2	52
3	24	2	38	2	52
3	24	2	38	2	52
3	24	2	38	2	52
3	24	2	38	2	52
3	24	2	38	2	62
3	24	2	38	1	62
3	24	2	38	2	65
2	24	2	38	2	65
2	24	2	38	2	65
2	24	2	38	1.5	65
2	24	2	38	1	65
2	24	5	42	1	65
2	24	5	42	1	65
2	24	4	42	2	73
2	24	4	52	2	73
2	24	4	52	2	73
3	30	4	52	2	73
2	30	4	52	1	73
2	30	4	52	1	73
2	30	3	52	1	73
2	30	3	52	1	73
2	30	3	52	2	80
2	30	3	52	1	80
2	34	3	52	1	80
2	34	3	52	1	80
2	34	3	52	1	80
1.5	34	3	52	1	80
1	34	3	52	1	95
1	34	3	52	1	95

Table 12, continued.

Reactive distance cm	turbidity NTU	Reactive distance cm	turbidity NTU	Reactive distance cm	turbidity NTU
4	38	3	52	1	95
4	38	3	52	1	95
4	38	3	52		
4	38	2	52		
4	38	2	52		
3	38	2	52		
3	38	2	52		
3	38	2	52		
3	38	2	52		
3	38	2	52		
3	38	2	52		
3	38	2	52		
3	38	2	52		

Table 13. Maximum reactive distance (cm) data for 135 mm juvenile Arctic grayling in water of various turbidities (NTU).

Maximum Reactive Distance cm	Turbidity NTU	Maximum Reactive Distance cm	Turbidity NTU
16	2.4	5	26
14	2.4	5	26
12	2.4	3	26
12	2.4	4	70
10	2.4	2	70
10	2.4	2	70
10	2.4	2	70
10	2.4	2	70
10	14	2	70
10	14	2	70
10	14	3	80
8	14	2	80
8	14	2	80
8	14	2	80
8	14	2	80
7	14	2	80
7	14	3	88
6	26	3	88
3	88		

QUALITY CONTROL

Accuracy and precision of non-filterable residue measurements related to USEPA method 160.2 (USEPA manual 600/4-79-020) was checked by analyzing 2 quality control samples sent by the U.S. Environmental Protection Agency, Environmental Monitoring and Support Laboratory, Cincinnati, Ohio. The true values of the quality control samples was 31.2 mg/L. The values I obtained were 32.7 mg/L and 30.9 mg/L. Although the higher of these values fall slightly outside the 95% confidence interval provided by the EPA, the standard deviation (1.27 mg/L) is less than the standard deviation provided by the EPA (1.84 mg/L), and the mean of the values is not significantly different from the true value ($p > 0.25$). The results of the quality control sample analysis indicate that the non-filterable residue measurements are accurate and precise.

The accuracy of total recoverable metals analysis done by Northern Testing Laboratories, Inc. (NTL) was checked as described on page 20. NTL's analysis was accurate for arsenic, copper and lead, but was slightly inaccurate for zinc and silver.

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